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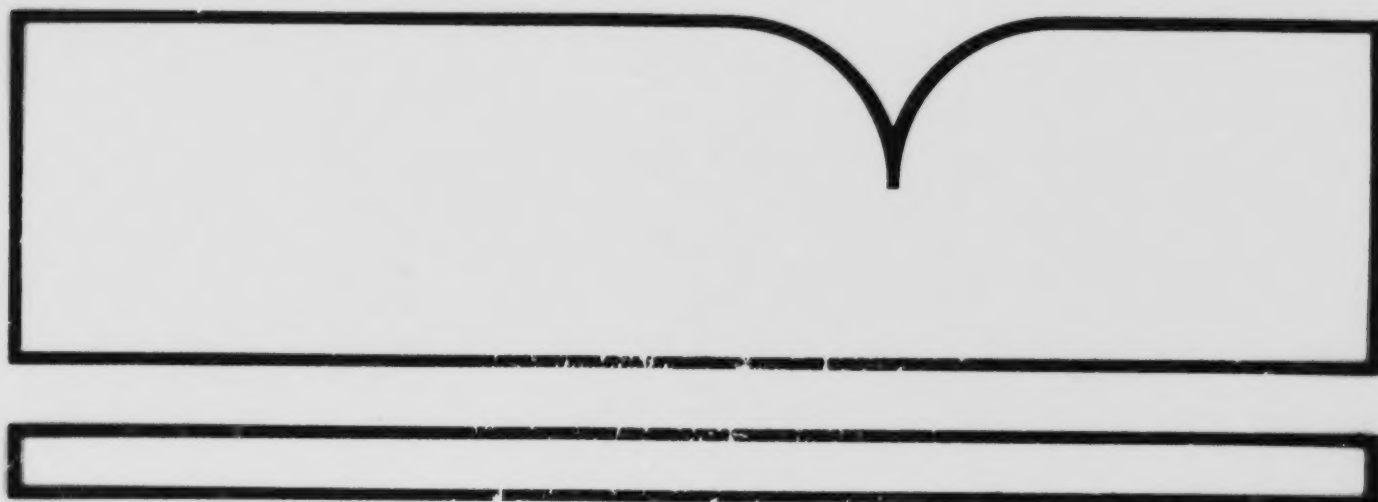
· Ecological Impacts of Sewage
Discharges on Coral Reef Communities

Tetra Tech, Inc., Bellevue, WA

Prepared for

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Ecological Impacts of Sewage Discharges on Coral Reef Communities

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ECOLOGICAL IMPACTS OF SEWAGE
DISCHARGES ON CORAL REEF COMMUNITIES

September, 1983

by

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Washington, D.C. 20460

EPA REVIEW NOTICE

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INTRODUCTION

Sewage pollution of coral reefs has been recognized as a major environmental problem for some time (e.g., Doty 1969; Banner 1974). Current U.S. environmental regulations allow Publicly Owned Treatment Works (POTWs) to apply for a modified National Pollutant Discharge Elimination System permit to discharge effluent receiving less-than-secondary treatment to marine waters. Under Section 301(h) of the 1977 Clean Water Act (as amended by the Municipal Wastewater Treatment Construction Grant Amendments of 1981, P.L. 97-117), POTWs are required to demonstrate to the U.S. Environmental Protection Agency that less-than-secondary treatment of their discharge will not result in certain adverse ecological impacts. The effects of effluent suspended solids and nutrients on coral reef communities are of special concern. In the context of the 301(h) sewage discharge evaluation program, coral reefs are considered "distinctive habitats of limited distribution." Their protection is especially important because of their ecological significance or direct value to man.

This report provides a synthesis of current information on the ecological impact of sewage discharges on coral reefs. Three major components of sewage pollution are addressed: 1) eutrophication associated with high nutrient concentrations in discharged wastewaters, 2) sedimentation of suspended solids, and 3) toxic effects. A review of sewage discharge impacts is presented, with emphasis on nutrient enrichment aspects (Section 2). The effects of solids deposition on corals are considered in Section 3. Finally, the available data are synthesized to develop functional relationships between discharge characteristics, sedimentation rates, and reef community impacts (Section 4).

The remainder of this introduction provides a brief summary of coral reef ecology, the susceptibility of reef communities to pollution impacts, and the potential for ecosystem recovery.

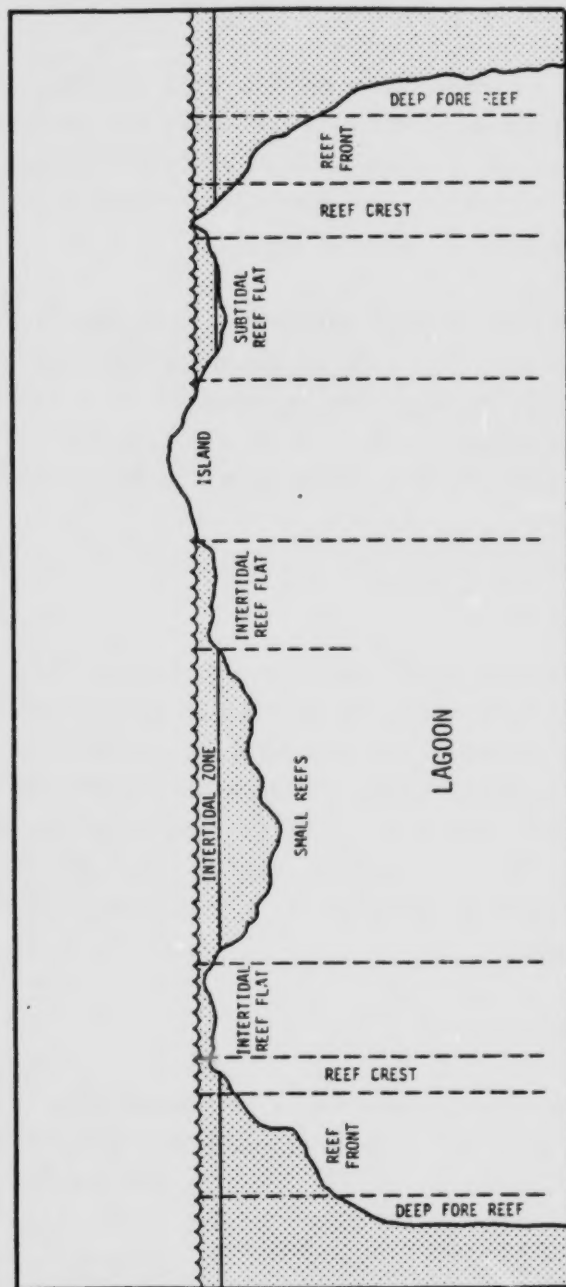
CORAL REEF ECOLOGY

Literature on the basic ecology of coral reefs is voluminous. Most of this information is reviewed in detail by Wells (1957), Stoddart (1969a), Sheppard (1982), and others. The intent of the following sections is to present brief descriptions of coral reef habitats, biological communities, and environmental controls of reef growth. The reader is referred to the reviews cited above and the specific papers cited below for further information on coral reef ecology.

Coral Reef Architecture and Habitats

Based on geomorphology and dominant biotic assemblages, complex classification schemes have been developed for coral reef types (Stoddart 1969a; Ladd 1977) and habitat zones (e.g., Wells 1957; van den Hoek et al., 1975; Sheppard 1980a). A simplified version of reef zonation includes the following characteristic habitats (Figure 1):

- Sandy beach - borders the reef proper; with calcium carbonate sediments, low diversity, no living corals
- Coral lagoon - several meters to greater than 50 m deep; varies from high cover and diversity of living corals to few scattered heads interspersed with sandy bottom and seagrass beds
- Reef flat - intertidal or shallow subtidal; usually high diversity and cover with branching, encrusting, and delicate corals in Caribbean; planer rock surfaces covered with algae and seagrass in Indo-Pacific
- Reef crest - supratidal to shallow subtidal; high wave energy zone on windward reefs; upper portions encompass algal ridge or coral rubble zone
- Reef front and terrace - subtidal zone with generally steep slope, complex topography, and large sediment transport; "groove and spur" zone; often high diversity and cover with



SOURCE: Modified from Demond, 1957

Figure 1. Generalized profile of a coral atoll.

massive staghorn, brain, and branching corals (many delicate gorgonians in Caribbean); deeper terraces may have scattered coral heads interspersed with deep sediments

- Seaward reef slope or deep fore reef - deep subtidal zone mostly beyond the influence of surface waves and currents; steep to nearly vertical slope; patches of coral rubble and sand; lower diversity, with zooxanthellae-containing corals and algae rare or absent at greater depths.

Depth ranges for the above habitats vary depending on exposure, reef type, geographic region, and water quality. As an example, the reef front of a typical coral reef extends from near the surface to 15 m (49 ft), the submarine terrace is found at about 15-18 m (49-59 ft), and the seaward reef slope occupies the 18-100 m (59-328 ft) depth zone (Stoddart 1969a; Goreau et al., 1979).

Biological Communities

The biological communities of coral reefs have been described in detail by many authors (e.g., see references in Jones and Endean 1973, 1976). Glynn (1973) reviewed reef communities and biotic interactions for the Western Atlantic region [also see Milliman (1973) and Stoddart (1976) on the Caribbean], and Maragos (1972) and Grigg (1983) described the ecology of Hawaiian reef corals. Other reviews include Wells (1957) and Wiens (1962) on marine biota of Indo-Pacific atolls, and Stoddart (1973) on reef communities of the Indian Ocean.

Corals and Associated Microflora--

The principal foundation species of coral reef communities, i.e., those species that contribute the most to community structure (Dayton 1972), are the hermatypic corals. Reef-building corals provide food and habitat for a wide variety of organisms (Robertson 1970; Patton 1976). All healthy hermatypic corals harbor symbiotic zooxanthellae (Dinophyceae), which apparently contribute substantially to coral nutrition (Muscattine 1973), reef calcification (Goreau et al., 1979), and primary production (Yonge 1972). Although corals feed extensively on zooplankton (Muscattine 1973),

they also consume planktonic algae, bacteria, and dissolved organic matter (Sorokin 1973a, b, c,). After death, the coral skeletons contribute to the reef framework and sedimentation processes (Stoddart 1969a).

Living corals produce copious amounts of mucus that is used in sediment rejection and prey capture (Hubbard and Pocock 1972; Lewis 1976). Eventual release of mucus from the surface of the coral colony results in the formation of organic aggregates (Johannes 1967). These organic aggregates are an energy-rich food source for small reef fishes and possibly zooplankton (Benson and Muscatine 1974; Ducklow and Mitchell 1979). The important trophic role of bacteria and detritus in coral reef systems is discussed by Johannes (1972), Sorokin (1973a, b, c), and Ducklow and Mitchell (1979).

Competition for space among reef corals has been studied in relation to light availability, colony growth rates, colony shape, and aggressive interactions. In general, fast-growing branching species (e.g., Acropora spp.) may kill smaller colonies indirectly by shading them (Connell 1973). Massive species in the families Mussidae, Meandrinidae, and Faviidae are able to inhibit growth of neighboring colonies by extracoelenteric digestion with mesenterial filaments (Lang 1973; Connell 1973). Lang (1973) established a competitive aggression hierarchy for western Atlantic species. Sheppard (1979, 1980a) reported on interspecific aggression, diversity, and depth zonation of reef corals of the Chagos Atolls (Central Indian Ocean). The complexity of interspecific interactions and the unpredictability of competitive outcomes in nature may limit the extent to which aggression hierarchies influence coral community structure (Bak et al., 1982).

The principal hermatypic species responsible for coral reef formation are listed in Table 1 (see Sheppard 1982 for references). In the Atlantic province, the main reef-forming species is the massive coral Montastrea annularis. Branching Acropora palmata occurs on seaward slopes and an encrusting Agaricia/Millepora assemblage occurs in shallow turbulent areas (Stoddart 1969a; Glynn 1973). In the Indo-Pacific region, Pocillopora and Acropora species are most often cited as primary framework builders. Acropora species are extremely rare in Hawaii (Grigg et al., 1981). There, Porites compressa, P. lobata, and Montipora verrucosa assume the role of dominant reef builders, especially on the well developed reefs of leeward

TABLE 1. PRIMARY FRAMEWORK BUILDERS OF CORAL REEFS

Species	Site (depth, m)
Atlantic Ocean	
<u>Acropora palmata</u>	Jamaica (0-6) Florida (0-3) Bonaire Lesser Antilles (0-10)
<u>A. cervicornis</u>	Bonaire
<u>Porites furcata</u>	Panama (0-2.5)
<u>Montastrea annularis</u>	Jamaica (6-20+) Lesser Antilles (6+)
<u>M. cavernosa</u>	Bonaire
<u>Millipora</u> spp.	Puerto Rico (8-20)
Coralline algae	Lesser Antilles (shallow)
Sclerosponges	Lesser Antilles (shallow) Jamaica (70-105)
Indo-Pacific Ocean	
<u>Pocillopora damicornis</u>	Panama (0.5-6)
<u>Pocillopora</u> spp.	Galapagos
<u>Stylophora pistillata</u>	Eilat (0-4)
<u>Acropora palifera</u>	Chagos (0-4)
<u>A. humilis</u>	N. Great Barrier Reef (0-5)
<u>A. hyacinthus</u>	Lord Howe Is.
<u>Galaxea astreata</u>	N. Great Barrier Reef (5-11)
<u>Porites californica</u>	Chagos lagoon (4-10)
<u>Millipora</u> spp.	Chagos lagoon (20-30)
Coralline algae	Cocos Is.
<u>Porites lobata</u> ,	Eilat (2)
<u>Porites compressa</u> , and	Hawaii and central Pacific
<u>Montipora</u> spp.	Atolls

Source: Sheppard (1982) and references therein.
Smith (17 April 1983, personal communication).

coasts (Maragos 1972; 1973; Grigg 1983). Coral reefs of the Indo-Pacific generally have a greater number of coral species (total of 700 species) than those of the Caribbean region (total of 60 species) (Milliman 1973). Hawaiian reefs are unusual in the Indo-Pacific region in having only 40 to 50 species of stony corals, possibly due to their geographic isolation (Grigg 1983).

Macrophytes--

The conspicuous macrophytes of coral reefs include coralline algae, fleshy algae, and seagrasses. The green alga Halimeda is an important primary producer on coral reefs throughout the tropics (Doty 1973; Hillis-Colinvaux 1980). Crustose coralline algae, such as Hydrolithon reinboldii in Hawaii and several species of Porolithon throughout the Indo-Pacific, contribute substantially to reef construction (Littler 1973). Porolithon and to a lesser extent Lithothamnium are the primary components of the algal ridge zone found at the crest of Indo-Pacific coral reefs (Stoddart 1969a). Other algae are agents of reef destruction, either directly by boring into coral skeletons (Highsmith 1981) or indirectly by overgrowing coral colonies (Banner 1974; Sammarco 1982).

Tropical seagrass beds are often found on reef flats and shallow sandy terraces adjacent to coral reefs. Seagrass beds serve as foraging grounds for reef fishes, sea urchins, sea turtles, and manatees; and as nursery grounds for commercial species such as the pink shrimp Penaeus duorarum, mullet, sea trout, and stone crab [Zieman (1975) and references therein]. Much of the organic production of seagrass and its associates supports detritus-based food webs. By acting as baffles against extreme wave action, seagrass beds also stabilize the substrate and limit sediment resuspension.

Grazers--

The conspicuous grazers of coral reefs include fishes, sea urchins, manatees, and sea turtles. The conspicuous lack of vegetation cover in many coral reef habitats can often be attributed to the efficient grazing activity of sea urchins and fishes (Ogden 1976; Hay 1981). On most reefs, the main herbivorous fishes are parrotfishes (Scaridae) and surgeon fishes (Acanthuridae) (Ogden 1976; Goldman and Talbot 1976), and perhaps damselfish (Pomacentridae) in shallow back-reef zones (Williams 1981).

Grazing by urchins and fishes prevents space monopolization by fleshy algae, which may otherwise overgrow living corals and coralline algae, trap sediments, and discourage recruitment of coral (Fishelson 1973; Birkeland 1977; Hay 1981, Sammarco 1982). Grazing on seagrasses by fish and urchins also maintains barren areas, or halos, separating reef borders and adjacent seagrass beds (Ogden et al., 1973; Tribble 1981). Algal biomass is usually greatest where grazing pressure is low, such as on shallow wave-washed platforms and in deeper areas of the fore reef (Ogden 1976; Benayahu and Loya 1977; van den Hoek et al., 1975).

The effects of sea urchins and fishes on coral community structure are complex, depending on grazer abundance and the particular species involved. For example, grazing by the urchin Echinometra viridis on a patch reef in Discovery Bay, Jamaica, was highly patchy (up to 50 individuals/m²), whereas the effects of the urchin Diadema antillarum were more uniformly distributed at equivalent or lower densities (Sammarco 1982). In the absence of all urchins, percent cover of adult corals was greatly reduced by algal overgrowth. However, removal of Diadema alone enhanced coral cover because this species removes juvenile corals (especially Agaricia and Porites) by its normal feeding activity. In a back-reef habitat of Discovery Bay, interference competition by three spot damselfish (Eupomacentrus planifrons) promotes coexistence of the two echinoids (D. antillarum and E. viridis) via habitat partitioning (Williams 1981). Effects of damselfish competition on urchin densities and distributions may prevent destructive overgrazing of the coral substrate by the echinoids. Finally, exclusion of herbivores from territories defended by small pomacentrid fishes may result in development of a thick algal mat, which restricts coral growth and diversity (Vine 1974; Potts 1977). In some instances, however, pomacentrid territoriality has been correlated with increased diversity of adult corals (Sammarco 1980). Wellington (1982) demonstrated that damselfish presence favors branching pocilloporid corals over massive coral species by protection of pocilloporids against grazing corallivores and by cultivation of algal mats on massive coral species.

Predators--

Predators and parasites on living corals include polychaetes, gastropods, asteroids, echinoids, crustaceans (cyclopoid copepods, cirripedes, and brachyuran crabs), and fishes (Robertson 1970; Glynn 1982). Of these, fish and asteroids (discussed below) are probably the most important. However, predation by the sea urchin Eucidaris on Pocillopora has been shown to limit the lateral and vertical expansion of Galapagos coral reefs (Glynn et al., 1979).

Scarids (parrotfishes) are probably the most important fish predators on coral populations (Bakus 1972; Frydl 1979), even though predation by scarids may be incidental to their grazing activity (S.V. Smith, 17 April 1983, personal communication). As with Diadema grazing, the effect of parrotfishes varies with predator density. At low grazing intensities, the reef community is dominated by fleshy algae. At intermediate intensities, a high diversity of corals is found; but with dense parrotfish populations, coral diversity and cover are low (Brock 1979). Other fishes known to feed directly on corals include species of triggerfish (Balistidae), filefish or leather-jackets (Monacanthidae), butterflyfish (Chaetodontidae), and a few damselfish (Pomacentridae) and small wrasses (Labridae) (Randall 1974; Birkeland and Neudecker 1981). Although fish predation on corals may be greatest in shallow waters above 10 m (33 ft) on some reefs (Bakus 1969, 1972), substantial feeding activity has been reported at depths to at least 65 m (213 ft) (van den Hoek et al., 1975). Control of Pocillopora depth distribution by fish predation has been suggested by field experiments (Neudecker 1977, 1979). Fish predation may be instrumental in maintaining the high diversity of reef corals and other invertebrates (Neudecker 1979; Ayal and Safriel 1982).

Further information on trophic roles of carnivorous reef fishes is available in the detailed reviews by Goldman and Talbot (1976) and Sale (1980). Whether the community patterns of coral reef fishes are determined by resource partitioning (Anderson et al., 1981) or stochastic events of recruitment and mortality (Sale and Williams 1982) is undetermined at present.

In the Indo-Pacific region, population increases of the asteroid Acanthaster planci are responsible for widespread mortality of corals (Endean 1973). On some reefs, Acanthaster may increase coral species diversity by compensatory mortality of dominant forms (Porter 1972; Colgan 1981), whereas in other areas preferential predation on rare coral species decreases diversity (Glynn 1974). Survival of Millepora communities, colonization of dead coral skeletons by algae, and invasions of alcyonarians have been noted following Acanthaster infestations (Sheppard 1982). Endean (1973; 1977) has maintained that Acanthaster outbreaks are probably triggered by extensive collection of molluscan and piscine predators of the starfish by humans. Other authors have suggested natural causes (Frankel 1977), such as unusually good recruitment of Acanthaster larvae under optimal conditions of food, temperature, salinity, and predation (Lucas 1973, 1982; Birkeland 1982).

Abiotic Controls on Coral Reef Development

The distribution and growth of coral reefs are influenced by several abiotic factors, including light availability, salinity, temperature, turbulence, sedimentation, and dessication. Because of their dependence on photosynthates produced by zooxanthellae, corals are generally limited by light availability to depths above 60-100 m (van den Hoek et al., 1975; Sheppard 1982). Experimental shading of a coral reef for 5 wk decreased primary production and caused death of corals (Rogers 1979). High turbidity combined with low salinities and siltation from runoff may also lead to mass mortalities of reef organisms, especially following periods of excessive rainfall (Banner 1968). Reef development may be limited by low salinity (e.g., 25-30 ppt) near river outflows or by high salinity (50-70 ppt) in arid regions (Wells 1957; Stoddart 1969a). Corals are notoriously sensitive to thermal variations, and low temperatures (or associated competition from macrophytes) limit the poleward distribution of most hermatypic species (Rosen 1981; Sheppard 1982; Johannes et al., 1983). Dessication stress on reef flats exposed at low tide may favor opportunistic coral species such as Stylophora pistillata (Loya 1976b, c).

Coral growth is usually greatest on the shallow reef front where waves and currents prevent excessive sedimentation. Although hydromechanical stresses limit branching coral species to quieter waters, these forms are

well adapted to heavy siltation rates (Chappell 1980; Sheppard 1982). Heavy sediment loads produced by dredging restrict reef growth and favor coral species able to reject settled materials efficiently (Dodge and Vaisnys 1977; Bak 1978).

A semiquantitative model proposed by Chappell (1980) integrates the effects of natural environmental "stresses" on coral growth forms, diversity, and reef development (Figure 2). Detailed reviews of ecological controls on coral populations are available in Stoddart (1969a) and Sheppard (1982).

SENSITIVITY TO POLLUTION

Coral reef ecosystems are considered extremely sensitive to environmental perturbations, including various forms of pollution (Johannes 1975; Loya and Rinkevich 1980). The high sensitivity of coral reefs to pollution stresses is linked to three factors (Johannes and Betzer 1975; Johnson and Pastorok 1982):

- Narrow physiological tolerance of corals
- Susceptibility of key species interactions to perturbation (e.g., plant-herbivore relationships, algae-coral competition)
- Increased effects of toxic pollutants at higher temperatures.

Corals have extremely narrow tolerance ranges for environmental conditions (Johannes 1975; Endean 1976). Thus, any variation of physical-chemical parameters outside their usual narrow range could be detrimental to coral growth and survival (Endean 1976; Pearson 1981). Destruction of hermatypic corals by pollution leads to the eventual demise of many reef species dependent on living corals for food, shelter, and refuge from predators (Johannes 1975). Through initial disruption of complex symbiotic relationships, pollution impacts may cascade throughout the reef system.

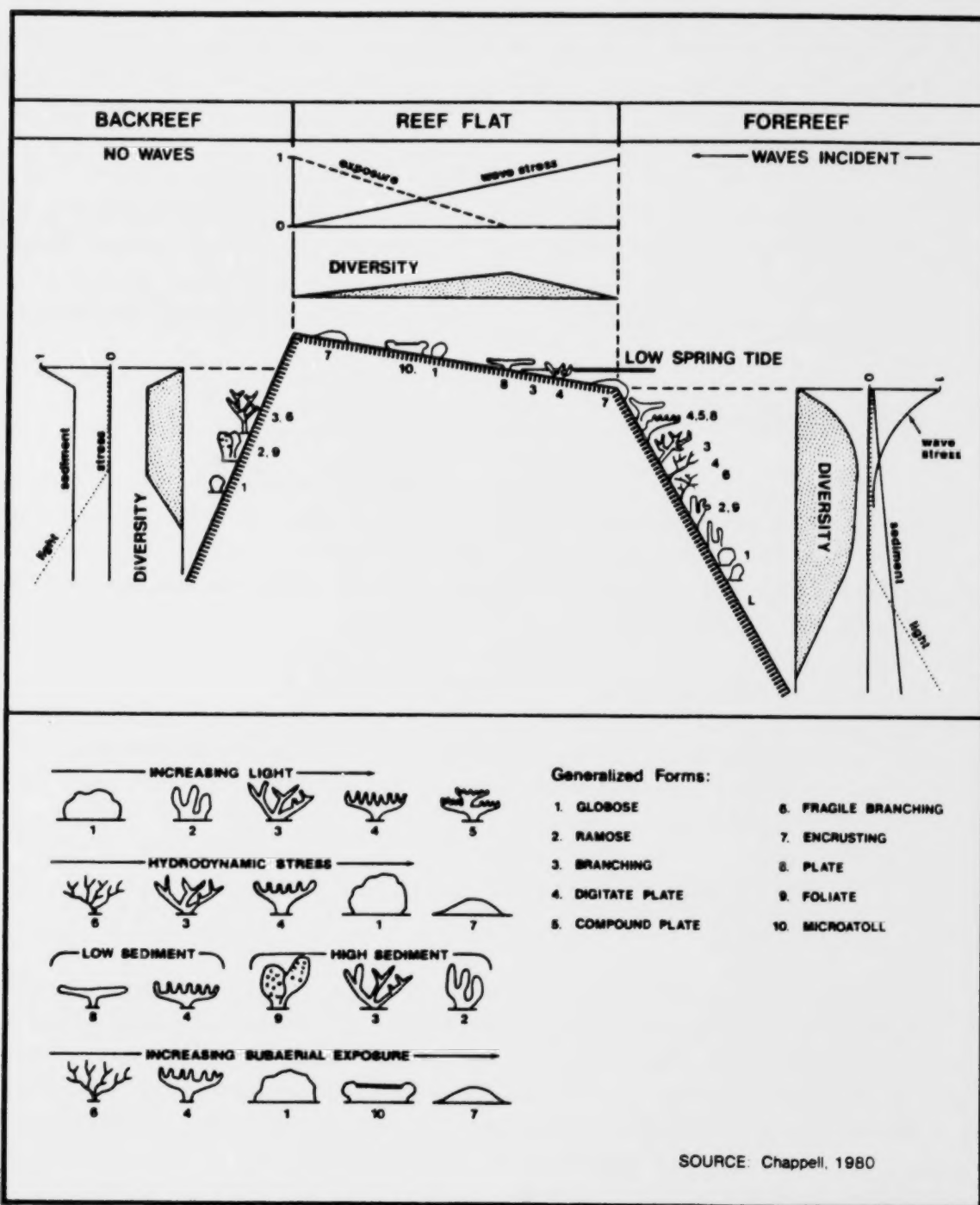


Figure 2. Idealized variation of coral growth forms, diversity, and abiotic factors across a simple reef profile.

Impacts of toxic pollution may be enhanced at high water temperatures common to coral reef environments (Johannes and Betzer 1975). Documented effects of high temperatures include increased solubility, faster biotic uptake, and greater toxicity of pollutants tested.

RECOVERY POTENTIAL

Recovery of coral reef communities from small, localized disturbances usually requires a decade or less (Table 2; Endean 1976; Pearson 1981). Small perturbations may increase coral diversity by compensatory mortality of dominant forms (Porter 1972; Connell 1978), while promoting asexual reproduction by fragmentation and dispersal of branching forms, e.g., Acropora palmata and A. cervicornis (Highsmith et al., 1980). Following severe damage, however, hermatypic corals may exhibit negligible recolonization, even after 20-30 years (Stephenson et al., 1958; Endean 1976). Extreme habitat modification may preclude complete recovery (Johannes 1975; Endean 1976; Pearson 1981).

Coral recolonization and rates of natural recovery are influenced by:

- Location of damaged habitat
- Size of disturbed patch
- Intensity and frequency of disturbance
- Reproductive "seed" population
- Larval and adult dispersal capabilities
- Current patterns
- Substrate available for larval settlement
- Larval/adult survival and competitive interactions
- Ecosystem productivity.

TABLE 2. ESTIMATES OF CORAL REEF RECOVERY TIME FOLLOWING DISTURBANCE

System	Disturbance		Measure of Recovery	Recovery Time	Reference
	Cause	Magnitude			
Reef slope--mixed growth forms, British Honduras	Hurricane	Large area Total destruction	Recolonization Normal cover	30 yr average 60-100 yr maximum	Stoddart 1963, 1969b, 1974
Reef slope--branched and staghorn corals, FL	Hurricanes	Many live corals remained	Visual appearance	5 yr 2 yr	Shinn 1976
Reef slope--mixed plate and massive corals, Great Barrier Reef, Australia	Hurricane	Major storm	Good coverage of well developed colonies	<20 yr	Woodhead (App. E in Walsh et al., 1971)
Reef slope--reefs on submerged lava, HI	Volcanic	Colonization of sterile habitat	Percent cover No. species Diversity	20 yr exposed 50 yr sheltered	Grigg and Maragos 1974
Reef slope--branching and massive corals, Persian Gulf	Cold	Acropora damage Massive corals unaffected	Recolonization Visual appearance	4 yr	Shinn 1976
Reef slope--Gulf of Mexico	Red tide	Most fish, corals, other invertebrates died	Recolonization of small colonies	~1 yr	Smith 1975
Reef flat--Guam	Low tide	Mass mortalities, including corals	Recolonization Normal cover	>3 yr	Yamaguchi 1975

TABLE 2. (Continued).

System	Disturbance		Measure of Recovery	Recovery Time	Reference
	Cause	Magnitude			
Reef flat--Red Sea	Low tide	80 to 90% coral mortality	Return to previous community structure	5-6 yr	Loya 1975, 1976b
Patch reef--tabular <u>Acropora</u> , Enewetak Atoll	Nuclear blast	Total mortality Skeletons left intact	Regrowth of dominant coral Diversity	>13 yr	Johannes 1975
Reef front, terrace, and slope--Guam	<u>Acanthaster</u>	30% decrease in number of species >50% decrease in coral cover	Percent cover Colony size Growth forms	20 yr subtidal terrace 31 yr seaward slope, possibly 100's yr	Randall 1973
Reef lagoon--Arno Atoll, Marshall Islands	<u>Acanthaster</u>	Most coral died	Recolonization of small colonies	3 yr	Branham 1973
Reef slope--Innisfail, Queensland	<u>Acanthaster</u>	Extensive coral mortalities	Percent cover, No. colonies, No. species	>8 yr	Pearson and Endeau 1969 Pearson 1974, 1981
Reef slope--Feather Reef, Queensland	<u>Acanthaster</u>	Extensive coral mortalities	Recruitment of young colonies Community structure	<6 yr >12 yr	Pearson and Endeau 1969 Pearson 1981

TABLE 2. (Continued).

System	Disturbance		Measure of Recovery	Recovery Time	Reference
	Cause	Magnitude			
Great Barrier Reef-- Australia	<u>Acanthaster</u>	Extensive coral mortalities	Community structure	20-40 yr	Endean 1971, 1973, 1976
Tanguisson Reef-- Guam	<u>Acanthaster</u>	Major reduction in species richness, density and cover of corals	Community structure	12 yr	Colgan 1981
Several coral reefs-- Various locations	Floods Volcanic water Dredging	Extensive often complete mortalities of corals	Recolonization of small colonies	No recovery >30-50 yr	Endean 1976 Pearson 1981

In general, recovery of coral reef communities may be fastest on the reef flat and the shallow reef front, and slowest in the coral lagoon and seaward reef slope zones (Pearson 1981; Johnson and Pastorok 1982). Life history strategies of dominant species and key trophic interactions influence the initial response to natural or anthropogenic perturbations as well as the direction and timing of recovery (Loya 1976b, c; Pearson 1981; Highsmith 1982; Porter et al., 1982). For example, recovery of exposed coral reefs which are dominated by early successional communities is usually more rapid than recovery of sheltered reefs where communities are closer to "climax" associations (Grigg and Maragos 1974; Table 2). Opportunistic species occupy disturbed habitats in lieu of "climax" community taxa (Loya 1976c). Finally, recovery of corals may be delayed if depression of grazer populations allows space monopolization by benthic algae (Fishelson 1973; Birkeland 1977; Sammarco 1982).

IMPACTS OF SEWAGE DISCHARGES

The effects of sewage discharges on coral reef communities are discussed in the following sections. After a brief discussion of potential impacts, a summary of case histories is presented.

POTENTIAL IMPACTS

The potential impacts of sewage effluent on coral reef communities may be broadly classified into four categories: 1) oxygen consumption, 2) nutrient enrichment, 3) sedimentation, and 4) toxicity.

Oxygen Consumption

Most tropical organisms, including corals, are living in environments near their critical tolerance levels for dissolved oxygen (Kinsey 1973). Hence, oxygen utilization by organic matter and microbes in sewage effluent "may constitute a significant stress" (Johannes 1975). Note that depression would be most critical at night, when oxygen levels are usually at their daily lows. Kinsey (1973) indicated that reef communities maintained a constant rate of oxygen consumption (respiration) as ambient dissolved oxygen levels declined to zero in field experiments. However, isolated corals may decrease their respiration rates to survive short periods of hypoxia (Yonge et al., 1932).

Nutrient Enrichment

Nutrient enrichment of coastal waters by sewage effluents can lead to dramatic modifications of physical, chemical, and biological parameters (Smith et al., 1981). Johannes (1975) summarizes the effects of anthropogenic nutrient inputs on reef communities and provides a discussion of "eutrophication-loving" organisms. Various aspects of enrichment are discussed more fully in later sections of this report.

Sedimentation

Sedimentation of sewage solids may result in accumulation of organic-rich deposits around outfalls located in poorly flushed waters (Maragos 1972). Resuspension of sediments and direct deposition of sewage particles on coral colonies could produce a variety of stresses including reduced light availability, enhanced bacterial populations, and the expenditure of additional energy to remove sediment from the colony surface (Roy and Smith 1971; Johannes 1975; Walker and Ormond 1982). Sedimentation impacts are treated in detail later (see below, Impacts of Sedimentation).

Toxicity

Toxic effects on corals and other reef organisms may result from chemicals present in sewage effluents (e.g., chlorine, phosphate, pesticides, PCBs, metals, and petroleum hydrocarbons) or release of hydrogen sulfide from anaerobic sediments. Although toxic substances contained in sewage effluents are potentially harmful to reef communities, their effects have received little investigation (but see, e.g., Davis 1971 as cited by Johannes 1975; Sorokin 1973c; Olafson 1978; Kinsey and Davies 1979; Loya and Rinkevich 1980).

CASE HISTORIES

The available case histories of sewage effects on coral reefs are summarized in Table 3. It is apparent that few detailed studies have been conducted.

Short-term studies conducted within a year after initiation of small sewage discharges reveal minor physical damage associated with outfall construction, but little evidence of coral community impacts has been found (e.g., Tsuda et al., 1975; Amesbury et al., 1976). Although algal populations may increase rapidly in response to sewage enrichment, community-level impacts on corals may require a year or more for development after initiation of the discharge. The most common response to high sewage loading is an increase in benthic algae and filter-feeding invertebrates (e.g., bryozoa, sponges, and tunicates), with a corresponding decrease in the diversity and abundance of hermatypic corals (Maragos 1972; Smith et

TABLE 3. CASE HISTORIES OF SEWAGE DISCHARGE IMPACTS ON CORAL REEFS

Discharge Characteristics						Reference
Location	Effluent Flow	Mass Emissions	Depth	Distance from Reef	Other	
Aqaba, Jordan Fringing reef Gulf of Aqaba, Red Sea	--	--	--	on reef	200-400 tons apatite per year. Primary treated sewage	Walker and Ormond (1982)
Pt. Gabert outfall Atoll lagoon reef Moer, Truk	0.0004- 0.0007 m ³ /sec (0.010 - 0.015 MGD)	--	8.4 m (27.6 ft)	on reef	Secondary treatment. Diffusers located on sand/silt substrate with little coral cover	Tsuda et al. (1975)
Tourist hotel outfalls, Fringing reefs, Northern coast, Jamaica	--	--	varied	many on reef	Varied treatment and suspended solids	Barnes (1973)
Donitsch outfall Lagoon fringing reef, Yap, M. Caroline Is.	--	--	5 m (20 ft)	on reef	Preoperational data only	Amesbury et al. (1976)
Maianae outfall Fringing reef Oahu, Hawaii	0.053 m ³ /sec (1.2 MGD)	204 kg/day annual average (450 lb/day)	10.4 m (34 ft)	on reef	Primary treatment. No industrial flow	Reed et al. (1977) Bowers (1979a) Tetra Tech (1982c)
Hilo outfall Patch reefs, Hilo Bay Oahu, Hawaii	0.131 m ³ /sec (3.0 MGD)	798 kg/day (1,760 lb/ day) end of permit	17 m (56 ft)	adjacent to reef outcrops	Primary treatment	Bowers (1979b) Tetra Tech (1982a)
Mokapu outfall Fringing reef Oahu, Hawaii	0.425 m ³ /sec (9.7 MGD)	1,175 kg/day (2,591 lb/ day)	27.1 - 33.2 m (89 - 109 ft)	on reef	Primary and trickling filter treatment. No industrial flow	Russo et al. (1977, 1979) Russo (1982) Tetra Tech (1982b)

Higher death rate of coral *Stylophora pistillata* in polluted area. Elevated algal and grazer abundance near outfall

No apparent effects on corals, algae, fish, and zooplankton^a

Possible impacts of detergents, nutrient enrichment, solids deposition, and bacterial contamination were noted^a

Damaged corals and seagrass beds due to outfall construction. Increased number of fish species in rubble and seagrass zones, decreased number of fish species in coral-dominated zones^a

No apparent effects on corals, micromolluscs, and zooplankton^a. Higher biomass of frondose algae and increased abundance of fish at outfall^a

Although coral cover near the outfall was lower than that at the control site, sewage was considered less important than other stressors

Decreased abundance of micromolluscs and coral from preoperational to postoperational period^{a,b}. Increased chlorophyll *a*, blue-green algae, and fish

TABLE 3. (Continued).

Kaneohe Bay outfalls, Lagoon fringing and patch reefs, Oahu, Hawaii	0.2-0.3 ^d m ³ /sec (5-8 MGD)	Similar to Mokapu outfall	7 m Marine Corps Station 8 m Kaneohe Municipal	Lagoon adjacent to reef	Prediversion and post- diversion data. Mainly secondary treatment. Two out- falls and nonpoint sources	Enhancement of chlorophyll a and zooplankton biomass. Decline of coral reefs due to overgrowth of benthic green algae and filter- feeders (bryozoans, sponges, tunicates, etc.)	Maragos (1972) Banner (1974) Caperon et al. (1976) Laws and Redalje (1979, 1982) Smith et al. (1981)
Sand Island outfall, Fringing reef, Oahu, Hawaii	2.98 ^c m ³ /sec (68.0 ^c MGD)	21,000 ^c kg/day (46,000 ^c lb/day)	11 m (35 ft)	on reef	Raw sewage discharge, 1955-1977	Complete absence of reef corals, enhancement of polychaete (Chaetopterus) populations within 400 m of outfall. Diversity of corals and other benthos enhanced in intermediate impact zone	Dollar (1980) Grigg (1975) Tetra Tech (1980)

^a Study design or survey methods places limitations on reliability of data.

^b Reduced percent cover of corals attributed to inconsistency in transect locations rather than sewage impact.

^c Values for new deep water outfall only.

^d End of operational period. See Figure 3 below.

al., 1981; Walker and Ormond 1982). In well-flushed waters along an open coast, few significant effects of sewage on coral reef communities have been demonstrated (Bowers 1979a, b; Russo et al., 1977, 1979; Russo 1982). However, the data from these latter studies on the response of corals is limited by improper study designs and small sample sizes.

Two case histories, the Gulf of Aqaba (Red Sea) and Kaneohe Bay (Hawaii) illustrate significant effects of sewage on coral reef communities. These studies are reviewed in detail below.

Gulf of Aqaba, Red Sea

The dynamics of coral mortality and algal growth resulting from sewage pollution have been studied on a fringing coral reef near Aqaba, Jordan (Walker and Ormond 1982). The reefs are potentially affected by a sewage discharge and by sediment deposition from an apatite ore loading facility. Although the relative importance of sewage pollution and apatite loss from ships was not determined, spatial effects of the sewage discharge were apparent. An increase in algal cover, a decrease in coral diversity, and an increase in small grazing molluscs were "obvious" from 5 m "upstream" to about 50 m "downstream" of the outfall. Walker and Ormond (1982) found that the death rate of coral tissue near the outfall was 4-5 times the death rate observed in a control area. Stylophora pistillata, a fast-growing opportunistic species (Loya 1976c), was the only remaining abundant coral species. Dead portions of colonies were covered with filamentous algae.

Although biomass of algae (mainly Ulva lactuca and Enteromorpha clathrata) was elevated at the outfall site compared with the control area, algal overgrowth did not appear to be a direct cause of coral death. Mortality was possibly related to inhibition of calcification by high phosphate concentrations, stress caused by high sediment loads, or localized bacterial infection triggered by the sewage effluent. Because grazer populations were higher at the sewage area compared to the control site, Walker and Ormond (1982) attributed the excessive algal growth to nutrient enrichment rather than a relaxation of grazing pressure. The authors concluded that the effect of increased sediment loads near the outfall were greatly aggravated by the ability of the algal mats to trap sediment, resulting in further stress to adjacent coral tissues.

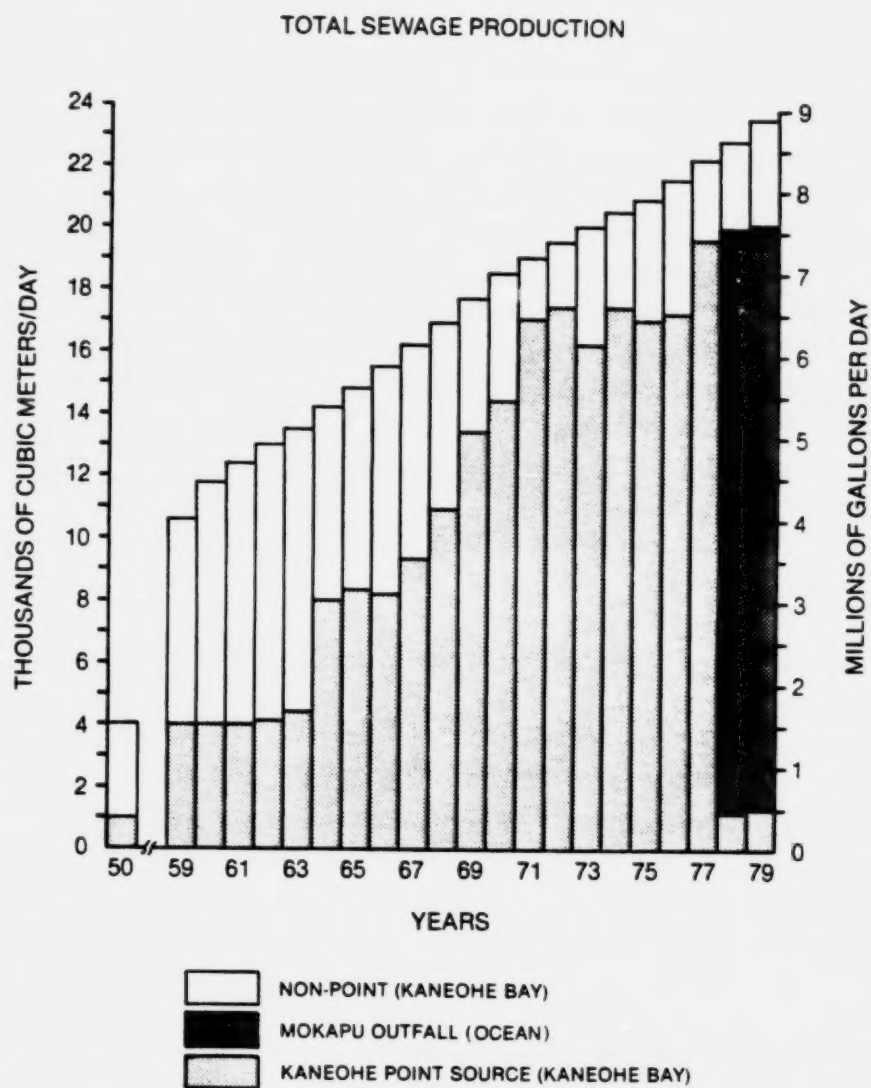
Kaneohe Bay, Hawaii

Kaneohe Bay, located on the northeastern side of Oahu in the Hawaiian Islands, received sewage inputs for a period of about 30 years (Laws 1981). After a period of increasing sewage discharge rates from 1950-1977, wastewaters from the Kailua-Kaneohe and Marine Corps treatment plants were diverted to an open ocean outfall off Mokapu Point (Figure 3). The effects of sewage enrichment on coral reef communities in Kaneohe Bay and their initial recovery following diversion of wastewaters are particularly well documented (e.g., Maragos and Chave 1973; Banner 1974; Laws and Redalje 1979, 1982; Smith et al., 1981). Although other impacts of urbanization such as dredging, increased runoff, and sedimentation complicated the initial interpretation of sewage discharge effects, (Banner 1974), the Kaneohe Bay diversion project offered an unprecedented opportunity for evaluation of sewage impacts on coral reef ecosystems (Smith et al., 1981). Biological changes following wastewater diversion are interpreted as a reversal of sewage impacts.

Physical Setting--

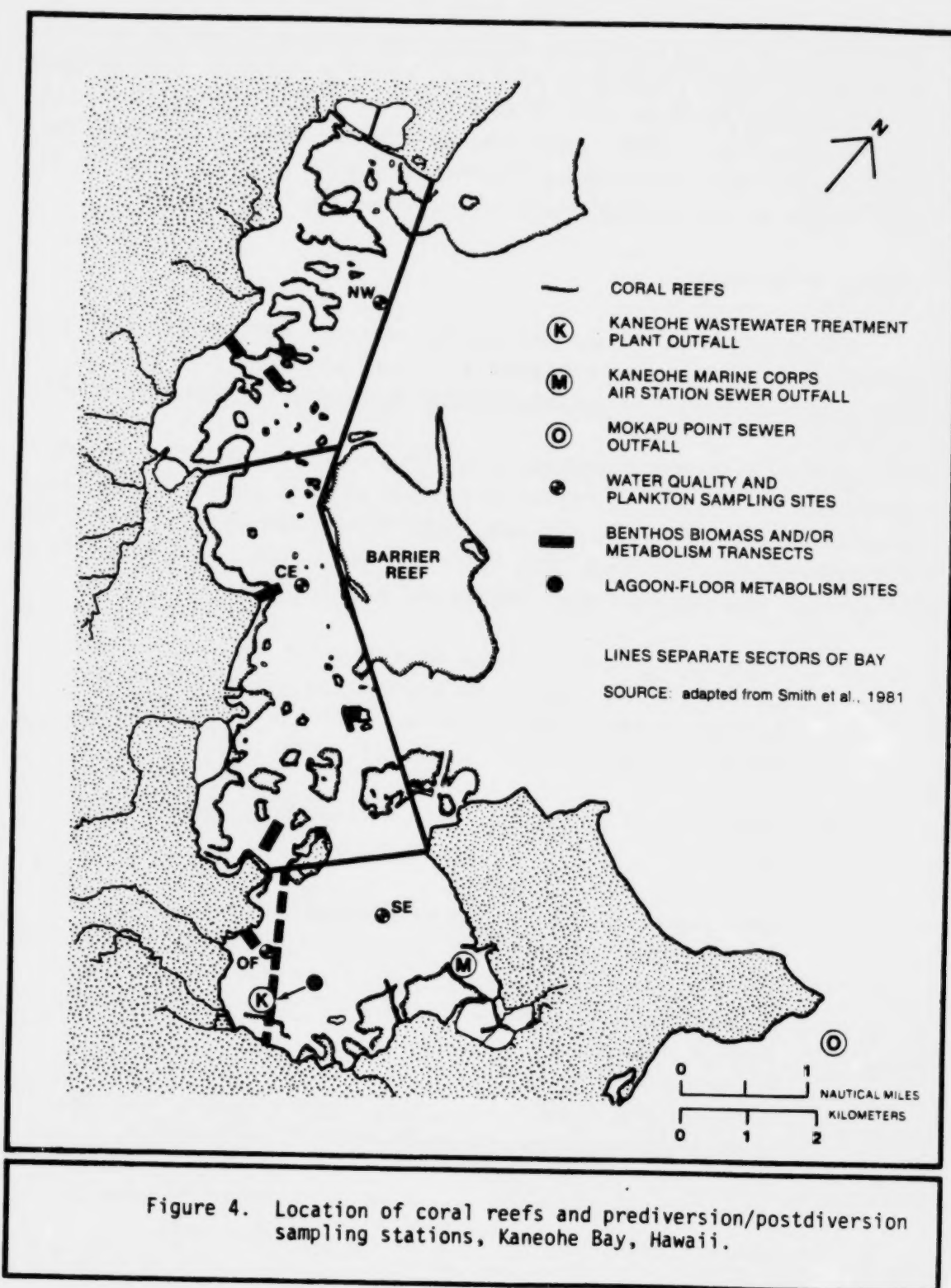
Kaneohe Bay is a semienclosed embayment, with a barrier reef extending along much of the bay mouth (Figure 4). The bay is divided longitudinally into four sectors on the basis of urban influence (especially sewage inputs), with increasing urbanization from north to south. The area of the inner bay is 31 km² (12 mi²), and the mean depth varies from 5.0 m (16.4 ft) in the outfall (OF) sector, where the two major sewage discharges were located, to 10.2 m (33.5 ft) in the southeast (SE) sector (Smith et al., 1981). A variety of coral reef and lagoon habitats are found in Kaneohe Bay (Smith et al., 1981).

Water enters the bay primarily across the broad barrier reef at the bay mouth, then moves into the southern basin from the central sector (Bathen 1968; Smith et al., 1981). In the inner central (CE) and northwest (NW) sectors, the predominant flow is from the south to the northwest. Water exits from the bay primarily through two deep channels, one north and one south of the barrier reef. Because the SE sector is largely isolated from direct oceanic exchange by Coconut Island and a system of shallow reefs,



SOURCE: Smith et al., 1981

Figure 3. Sewage loading in Kaneohe Bay and Kailua Bay, Hawaii, 1950-1979.



water circulation is restricted compared to the CE and NW sectors. Nevertheless, tidal flushing is efficient, yielding residence times of about 24 days for the SE sector and 12 days for the rest of the bay (Sunn, Low, Tom, and Hara, Inc., 1976). From a model of tidal flushing in Kaneohe Bay, Smith et al. (1981) calculated a flushing time of 8 days for the entire bay and 13 days for the southern basin (SE and OF sectors combined).

Impacts of Sewage--

The quantity of sewage wastewater entering Kaneohe Bay from 1950 through 1979 is shown in Figure 3. From 1963 on, the bulk of the wastewaters discharged to the bay resulted from secondary sewage treatment.

Water Quality and Plankton--About 74 percent of the nitrogen and 75 percent of the phosphorus influx to Kaneohe Bay was contributed by sewage discharges from the Kaneohe municipal and Kaneohe Marine Corps Air Station treatment plants (Sunn, Low, Tom, and Hara, Inc., 1976). The main effects of nutrient enrichment on water quality and plankton were:

- Increased concentrations of inorganic phosphorus and particulate nitrogen: the average concentration of each nutrient in the outfall sector was almost four times that in the rest of the bay (Laws and Redalje 1979).
- Reduced water clarity (Laws and Redalje 1979; Smith et al., 1981).
- Enhancement of phytoplankton biomass and total primary productivity: e.g., average chlorophyll a near the outfall was 6.7 times the average value for the central and northwest sectors combined (Caperon et al., 1976; Laws and Redalje 1979; Smith et al., 1981).
- Appearance of "red tides:" e.g., Exuviella sp. near the outfall and throughout the south sector (Clutter 1972).
- Minor enhancement of zooplankton biomass, with a decrease in species diversity (Clutter 1972; Smith et al., 1981).

Other parameters showed little change with distance from the sewage outfalls or little response to sewage diversion. For example, average dissolved oxygen concentrations for the period January, 1976, to June, 1977, were similar throughout Kaneohe Bay, with the lowest time-averaged value of 94 percent saturation in the outfall sector (Smith et al., 1981). Inorganic nitrogen levels were insensitive to sewage loading because of rapid uptake in a nitrogen-limited system (Laws and Redalje 1979, 1982; Smith et al., 1981).

Benthic Invertebrates and Macrophytes--The primary impacts of sewage on benthic communities in Kaneohe Bay included:

- Decreased coral cover, taxonomic richness, and net calcification rate (Banner 1974; Kinsey 1979; Smith et al., 1981).
- Increased biomass of algae Ulva and Hydroclathrus in OF/SE sectors and bubble algae Dictyosphaeria cavernosa throughout the CE sector (Soegiarto 1973; Banner 1974; Smith et al., 1981).
- Shift of benthic community structure away from corals and associates toward filter feeders such as sponges and zoanthids (Maragos 1972; Smith et al., 1981)

In addition, the diversity of benthic algae in the southern basin may have been reduced, as indicated by the absence of common lagoon algae (Soegiarto 1973). Despite this trend and the spatial gradient of increasing coral species richness away from the south basin (Maragos 1973; Maragos and Chave 1973), Smith et al. (1981) concluded that sewage inputs did not cause major shifts in species composition of the benthos.

The effects of nutrient loading on coral communities in Kaneohe Bay can be attributed to several mechanisms (Maragos and Chave 1973; Laws 1981; Smith et al., 1981). First, abundant phytoplankton in the sewage enriched waters of the southern basin reduced light availability to coral zooxanthellae, probably resulting in poor nutrition, growth, and survival of

corals. Second, enhanced production of organic particles favored benthic filter-feeders. Living corals transplanted to the south bay were quickly outcompeted by tunicates, bryozoans, sponges, sabellids and other filter-feeders (Maragos 1972). Third, the low diversity of corals and other benthos in southern Kaneohe Bay was attributed in part to the toxicity of hydrogen sulfide in anoxic bottom sediments (Maragos and Chave 1973). Maragos (1972) found that the survival time of transplanted corals was directly proportional to distance from the sewage outfalls. Moreover, Sorokin (1973c) indicated that sulfide levels increased and the anaerobic layer expanded closer to the Kaneohe sewage outfall.

Finally, sewage influenced the coral reefs of central Kaneohe Bay indirectly by stimulating growth of Dictyosphaeria cavernosa (Smith et al., 1981). This green bubble algae kills corals (mostly Porites compressa) by forming thick mats which smother all underlying reef organisms (Maragos and Chave 1973; Banner 1974). As a result of grazing pressure and better flushing in the northwest sector, D. cavernosa failed to dominate benthic communities there. Bubble algae were rare in the southern basin (Soegiarto 1973), probably for the same reasons corals are absent (Maragos and Chave 1973). Smith et al. (1981) presented experimental evidence that light limitation was one important factor restricting growth of D. cavernosa in the outfall (OF) sector.

In the lagoon area of Kaneohe Bay, Maragos (1972) calculated that 8.5 percent of the living reef front was killed by the direct effects of sewage discharge, 23.5 percent was overgrown by Dictyosphaeria cavernosa, 29.3 percent was removed by dredging, and 9.8 percent was destroyed by freshwater influx and sedimentation. Further, he estimated that 26.4 percent of the reef fronts in the northern sector have died, 86.8 percent in the central sector, and 99.9 percent in the southern sector.

Fishes--The fish communities on reef slopes and crests exhibit substantial differences among the various sectors of Kaneohe Bay. Species richness is lowest in the southern basin (6 resident species) compared with the central (43 resident species) and northern sectors (40 resident species) (Smith et al., 1981). Presumably, this pattern results from the reduced habitat complexity of the southern sector due to the absence of living corals. In contrast, the species richness of fishes associated with the lagoon floor does not vary among sectors.

Planktivorous fishes (e.g., Stolephorus purpureus and Pranesus insularum) were more abundant in the southern basin than in the rest of the bay (Clarke 1973), perhaps due to enhancement of prey populations near the sewage outfalls. By the late 1970s, planktivorous species dominated fish communities of the central sector (Brock et al., 1979). In contrast, other reefs are generally dominated by predators on larger prey (Goldman and Talbot 1976). Some common reef dwellers (e.g., Stethojulis, Scarus, Acanthurus, and Chaetodon spp.) were rare in the southern sector (Key 1973).

Ecosystem Response to Sewage Diversion--

Diversion of sewage from Kaneohe Bay outfalls to the Mokapu outfall occurred in two stages: 1) the diversion of the Kaneohe discharge in December, 1977 and 2) the diversion of the Marine Corps discharge in May, 1978. The study by Smith et al. (1981) spanned a prediversion period (January 1976 through November 1977) and a "postdiversion" period (December 1977 through August 1979).

The responses of physical, chemical, and biological parameters to sewage diversion in Kaneohe Bay are summarized in Table 4 and Figure 5. Smith et al. (1981) evaluated the responses of biomass, nutrient concentrations, and rate parameters quantitatively as the percent decrease in the average value of each parameter for the southern basin (i.e., sector nearest the outfall) from the prediversion survey to the postdiversion survey. Water column variables were volume-averaged (Smith et al., 1981). A spatial gradient for the postdiversion period was determined as the ratio of a mean parameter value at Station OF to an average transition zone value (i.e., volume- or area-weighted means for stations NW and CE). Since detailed taxonomic identifications were not performed, changes in community structure were evaluated subjectively (e.g., Table 4).

Parameter responses to sewage diversion fall into three groups (Figure 5):

1. Moderate Response to Sewage Diversion, Moderate Spatial Gradient:

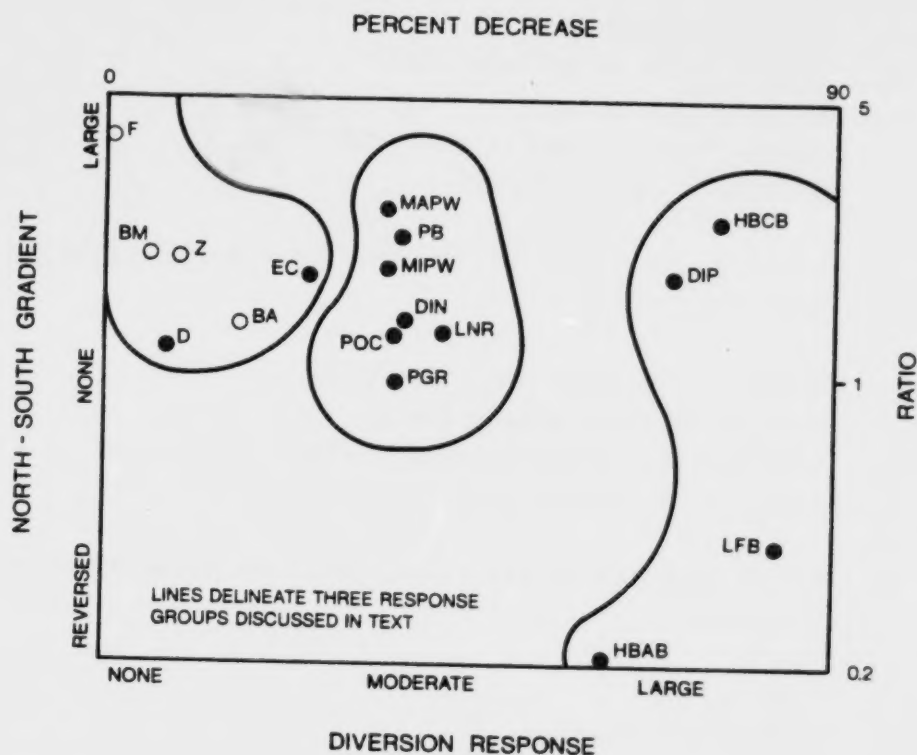
TABLE 4. SUMMARY OF RESPONSES TO SEWAGE DIVERSION,
KANEHOE BAY, HAWAII

Variable	Diversion Response ^a (Percent)	Spatial Gradient Response ^b
<u>Quantitative Variables</u>		
Dissolved inorganic nitrogen	37	1.4
Dissolved inorganic phosphorus	70	1.8
Particulate organic carbon	36	1.3
Phytoplankton biomass	37	2.3
Phytoplankton growth rate	36	1.0
Macroplankton dry weight	35	2.7
Microplankton ash-free dry weight	35	1.9
Hard bottom algal biomass	62	0.2
Hard bottom cryptofaunal biomass	76	2.5
Lagoon floor biomass	83	0.4
Lagoon floor nitrogen release	42	1.3
Extinction coefficient	25	1.8
Detritus	8	1.2
<u>Qualitative Variables:</u>		
<u>Community Structure</u>		
Zooplankton	slight	moderate
Benthic algae	slight to moderate	slight
Benthic macrofauna	slight or none	moderate
Fish	none?	large

^a Percent decrease in the southern basin (volume-weighted means of the southeast and outfall sectors of the bay).

^b Ratio of post diversion values in the outfall sector to values in the transition zone (volume- or area-weighted means of central and northwest sectors).

Source: Smith et al. (1981).



See footnotes to Table 4

SOURCE: Smith et al., 1981 (modified)

Figure 5. Spatial gradient and parameter responses to sewage diversion, Kaneohe Bay, Hawaii.

Dissolved inorganic nitrogen (DIN)
Particulate organic carbon (POC)
Phytoplankton biomass (PB)
Phytoplankton growth rate (PGR)
Macroplankton dry weight (MAPW)
Lagoon floor nitrogen release (LNR)
Extinction coefficient (EC)

2. Large Response to Diversion, Wide Range of Spatial Gradients:

Dissolved inorganic phosphorus (DIP)
Hard bottom algal biomass (HBAB)
Hard bottom cryptofaunal biomass (HBCB)
Lagoon floor biomass (LFB)

3. Slight Response to Diversion, Small to Large North-South Gradient:

Detritus (D)
Zooplankton structure (Z)
Benthic algal structure (BA)
Benthic macrofauna structure (BM)
Fish structure (F).

Variables in Group 1 include all of the water column mass parameters (except dissolved inorganic phosphorus). The various particulate and dissolved inorganic materials, phytoplankton, and microheterotroph masses in Group 1 exhibited gradients from high concentrations near the outfall to low values in the northern basin (Smith et al., 1981). These parameters decreased rapidly within the 1-2 yr following sewage diversion, possibly declining to baseline values (Smith et al., 1981). However, Laws (1981) and Laws and Redalje (1982) indicated that nutrients released from organic-rich sediments may have continued to support chlorophyll a biomass and nutrient levels above baseline values throughout the postdiversion period.

The second group of variables, including dissolved inorganic phosphorus and several benthic biomass measures, showed a large response to sewage

diversion (Table 4; Figure 5). Because the Kaneohe Bay system was far from phosphorus limitation before sewage diversion, the response of phosphate was large (Smith et al., 1981). The large response of benthic biomass variables to diversion reflected the decreased influx of particulate matter to the benthos compared with that contributed from a water column previously enriched with sewage nutrients. However, Smith et al., (1981) reported that benthic conditions at the end of their postdiversion survey did not match the "presewage" baseline. This result was attributed in part to the destruction or burial of hard substrate habitat by both "sewage-mediated biological activity" (e.g., overgrowth of corals by bubble algae) and heavy siltation from urban runoff. Moreover, the slow recruitment processes necessary for re-establishment of coral populations (see above, INTRODUCTION, Recovery Potential) precludes rapid succession and recovery of reef communities. By mid 1981, partial recovery of the coral communities was evident (Smith et al., 1981).

The third group of variables showed little response to sewage diversion. In Kaneohe Bay, the detrital pool was maintained by runoff and sediment resuspension, not by nutrient input from sewage. The slight change in community structure variables from prediversion to postdiversion periods may indicate that the effects of sewage inputs were minor. Alternatively, the benthic habitat may not have had sufficient time to return to a pre-discharge condition (Smith et al., 1981). Disintegration of reef rock, which was linked to shifts in reef community metabolism caused by the discharge, may account for the delay in recovery.

IMPACTS OF SEDIMENTATION

Quantitative measurements of solids deposition rates are generally unavailable for sewage discharges in coral reef environments. Consequently, the following discussion emphasizes sedimentation of particulate matter derived from terrestrial runoff, dredging and filling activities, and resuspension of bottom sediments.

POTENTIAL IMPACTS

The effects of sediment inputs to a coral reef environment depend primarily upon the relative sensitivities of the primary framework species.

Sensitivity of Corals to Sedimentation

The sensitivity of a coral species to rapid sedimentation depends on the sediment-trapping properties of the colony and the ability of individual polyps to reject settled materials. Horizontal plate-like colonies and massive growth forms present a large, stable surface for interception and retention of settling solids. Conversely, vertical plates and upright branching forms are less likely to retain sediments on the surface of the colony (Bak and Elgershuizen 1976; Dryer and Logan 1978; also see above Figure 2). Tall polyps and convex colonies are also less susceptible to sediment accumulation than are other growth forms (Lasker 1980).

The physical and chemical characteristics of the sediment are also important determinants of its effects on reef biota. Physical characteristics such as density and grain size composition may influence clearing rates within a given species. Sediment chemistry is also critical: sediments having high organic content, high BOD, or adsorbed toxic substances (e.g., pesticides) may exert more pronounced effects on the behavior and physiology of reef organisms than would less chemically active sediments (e.g., calcium carbonate, quartz) of the same grain size composition.

Lastly, sensitivity of corals to sedimentation may be influenced by the frequency of exposure to suspended solids, and the light attenuation which results from decreased water clarity. The occasional dosing of corals with sediments advected into the reef area (primary sedimentation) may be less harmful than repetitive dosing with resuspended sediments. Aside from differences in energy expended to clear sediments, rapid light attenuation with depth, which accompanies turbid conditions, may also affect photosynthesis by the zooxanthellae. This effect may be most pronounced in relatively clear waters, where the addition of a small amount of suspended material greatly decreases light transmittance. In systems where ambient turbidity is relatively high, a small increase in suspended material will have little effect on light transmittance.

Sediment Rejection--

Coral polyps reject sediment landing on the surface of the colony by four mechanisms: 1) polyp distension by uptake of water through the stomodeum, 2) tentacular movements, 3) ciliary action, and 4) mucus production (Marshall and Orr 1931; Hubbard and Pocock 1972). Although the efficiency of sediment rejection has been related to skeletal geometry and polyp morphology (Hubbard and Pocock 1972; Loya 1976a), polyp behavior may supercede morphological differences (Bak and Elgershuizen 1976).

Several factors limit the ability of corals to reject sediment. First, most coral colonies are unable to coordinate transport of sediment off the colony by the shortest possible route. The pathway followed by a sediment particle during the rejection process approximates a random walk (Dodge and Vaisnys 1977). Thus, the sediment rejection process is much more efficient in small or young corals than in large or old colonies. Second, silt is the largest particle size effectively removed by many coral species (Hubbard and Pocock 1972). Larger size fractions, which are removed by some species but not others, must be transported by polyp distension rather than relatively weak ciliary action.

Relative Species Sensitivity--

The relative sensitivities of some coral species to sedimentation effects are given in Table 5. Only those species which have a clear sensitivity ranking have been included in the table. The relative sensitivity of a few genera is given to illustrate the variation in rank among species within a genus (e.g., the common genera Acropora, Porites, Pocillopora). In general, coral species inhabiting the seaward margins of a reef are less tolerant of high sediment loads than species found in nearshore areas (Vaughan 1916; Marshall and Orr 1931; Stern and Stickle 1978).

Montastrea cavernosa is able to effectively reject sediments at deposition rates as high as $7-8 \text{ mg cm}^{-2} \text{ day}^{-2}$ (Lasker 1980). The upper limit to M. cavernosa's ability to remove sediment effectively appears to be about $14 \text{ mg cm}^{-2} \text{ day}^{-1}$ or slightly higher (Loya 1976a, Lasker 1980). Since M. cavernosa is an efficient sediment rejector and is commonly found in environments dominated by rapid sediment accumulation, many other coral species are less tolerable of heavy sediment inputs.

Individual Effects

High concentrations of suspended solids and rapid sedimentation are responsible for decreased coral growth rates, changes in colony growth form, and possibly increased mortalities.

Lethality--

A heavy coating of sediments or complete burial for more than several hours kills most corals (Edmondson 1928; Marshall and Orr 1931; Roy and Smith 1971).

Growth Inhibition--

Adverse effects of heavy sediment loads on coral growth may result from decreased light availability, abrasion, and energy expenditure for sediment rejection. High turbidity interferes with light penetration to the bottom and thereby limits photosynthesis of zooxanthellae and coral growth (Roy and

TABLE 5. SENSITIVITY OF SOME COMMON CORAL SPECIES TO SEDIMENTATION

Species	Sensitivity		
	Low	Moderate	High
<u>Montastrea cavernosa</u>	x		
<u>Siderastrea radians</u>	x		
<u>Siderastrea siderea</u>	x		
<u>Manicina areolata</u>	x		
<u>Fungia spp.</u>	x		
<u>Agaricia agaricites</u>		x	
<u>Acropora hyacinthus</u>			x
<u>Acropora corymbosa</u>			x
<u>Acropora cervicornis</u>			x
<u>Other Acropora spp.</u>		x	
<u>Porites astreoides</u>			x
<u>Other Porites spp.</u>	x	x	
<u>Pocillopora spp.</u>	x	x	

Source: Edmondson (1928), Marshall and Orr (1931), Yonge (1935), Hubbard and Pocock (1972), Ott (1975), Bak and Elgershuzien (1976), Loya (1976a), Bak (1978), Lasker (1980).

Smith 1971; Maragos 1972; Dallmeyer et al., 1982). Experimental additions of peat particles to field enclosures decreased primary production and reduced chlorophyll content by 22 percent in Montastrea annularis, indicating loss of zooanthellae from stressed corals (Dallmeyer et al., 1982). Calcification rates have been reduced by as much as 40 percent by natural resuspension of sediments (Dodge et al., 1974) and by as much as 33 percent by short-term dredging impacts (Bak 1978). Although some coral species (e.g., Montastrea cavernosa, Siderastrea siderea) are able to survive highly turbid conditions (Roy and Smith 1971; Loya 1976a; Randall and Birkeland 1978), reduction of light levels below the critical compensation point for photosynthesis may lead to cessation of growth and eventual death, especially in deepwater corals (Johannes 1975).

Abrasion of coral surfaces by suspended particulates may also contribute to decreased growth (Johannes 1975 and references therein; Loya 1976a). Wiens (1962) concluded that mechanical scour plays a role in the destruction of coral reefs, particularly at the reef margins.

Rejection of sediments by corals is an energetically-expensive process. Dallmeyer et al. (1982) demonstrated a significant increase in coral respiration rates during vigorous sediment-cleansing activities. Energy required for sediment removal is diverted from other metabolic functions, possibly leading to reduced growth and lower reproductive output (Aller and Dodge 1974; Dodge and Vaisnys 1977; Dallmeyer et al., 1982). Moreover, if coral polyps are occupied with sediment rejection activities, they may be unable to capture zooplankton effectively.

Growth Form Changes--

Aside from its effects on the rate of coral growth, rapid sedimentation is expected to produce changes in the growth form of coral colonies. Differential accumulation of sediment across the surface of a colony influences topographical variations in growth rate, possibly modifying colony growth towards forms more resistant to sedimentation (Marshall and Orr 1931; Roy and Smith 1971).

Population and Community Effects

Adverse impacts of sedimentation on coral populations and communities are evidenced by changes in the relative abundances of coral species, reduced substrate cover by living corals, lowered species diversity, and inhibition of larval recruitment.

Disturbance of Community Structure--

Healthy coral communities may exist in chronically turbid waters, especially when strong currents prevent the build-up of sediments on coral surfaces (Marshall and Orr 1931; Roy and Smith 1971). Nevertheless, corals exposed to high turbidity or rapid sedimentation are less diverse and less abundant than those at reference sites with clear water (Roy and Smith 1971; Loya 1976a; Randall and Birkeland 1978). Since coral depth distribution is limited by light penetration, increased turbidity levels may produce an apparent shift of coral depth distributions toward shallower waters. Because the relationship of species distributions to light levels is complex (Sheppard 1982), a simple response to elevated turbidity is unlikely.

Reduced Coral Recruitment--

Unconsolidated substrates are unsuitable for settlement and survival of some coral recruits (Harrigan 1972 as cited by Johannes 1975). For many coral species, accumulation of sediment over a hard substrate inhibits larval settlement and juvenile development (Edmondson 1928; Maragos 1972; Dodge and Vaisnys 1977).

CASE HISTORIES

A summary of case history data on sedimentation in reef environments is presented in Table 6. Only those studies containing quantitative estimates of natural sedimentation rates or anthropogenic inputs are included in the table. Qualitative information on sediment loading and response of corals is discussed later.

Examination of the data in Table 5 reveals that several investigators have found inverse correlations between sedimentation rates and coral

TABLE 6. CASE HISTORIES OF SEDIMENT IMPACTS ON CORAL REEFS

Location	Sediment Source	Sedimentation Rates ^a mg cm ⁻² day ⁻¹	Responses or Characteristics	Notes	Reference
St. Thomas, U.S.V.I. Fringing reefs in several bays and near small off-shore islands	Airport runway construction	0.7-5.9	No apparent effects on seagrass, corals, algae, and fish	10-cm diameter sediment traps	Rogers (1982)
Galea Island, Panama Nalu Nega, San Blas Is. Fringing reefs	Natural sources	0.3-0.9	No apparent impact on the coral <u>Montastrea cavernosa</u>	Measured sediment accumulation on dead coral	Lasker (1980)
Discovery Bay, Jamaica Lagoon fringing reefs	Resuspension	0.45-1.10	Coral growth was inversely related to sediment resuspension rate		Dodge et al. (1974) Aller and Dodge (1974)
Guayanilla Bay, Puerto Rico Fringing reefs	Resuspension by ship traffic	1.1-9.8	Low cover and diversity of corals related to high resuspension rates	Preliminary results	Morelock et al. (1979)
Barbados, West Indies W. Coast barrier reef	Resuspension	1-15	Percent dead coral tissue correlated with sedimentation rate on outer reef slope ^b		Ott (1975)
Negro Bank, Puerto Rico Fringing reefs	Rivers and re-suspension	East Reef 3.0 West Reef 15.0	High turbidity and sedimentation resulted in low cover and diversity of corals	Turbidity (FTU) East Reef 1.5 West Reef 5.5	Loya (1976a)
Key Largo, Florida Keys Patch reef	Dredging, 0.7 km (0.5 mi) upcurrent from reef	37 (6-125)	No apparent effects on resistant species of coral, <u>Siderastrea siderea</u>	Minor impact of dredging on reef sedimentation	Griffin (1974)
Puerto Rico	Engineering experiment	150	No apparent effects on seven coral species after 9 day's exposure ^b		Kolehmainen (1974)
Fouha Bay and Ylig Bay, Guam Fringing reefs	Rivers and resuspension	6-228	Higher species richness, percent cover, and colony size of corals near bay mouths where sedimentation was less		Randall and Birkeland (1978)
Kaneohe Bay, Hawaii Fringing reefs	Sewage discharge and watershed urbanization	36-41,096	Growth rate of corals correlated primarily with light intensity, not sedimentation rate	Short-term sediment trapping during winter storms produced high sedimentation rates	Maragos (1972)

^a Sedimentation rates include natural background values where anthropogenic sources are involved.^b Limited data available.

community parameters (e.g., species richness, abundance, growth). However, sedimentation rates from different studies listed in Table 5 can be compared only in an approximate manner for two reasons. First, variations in the design and deployment of sediment traps influence the absolute differences among sedimentation rates reported from different studies (cf. Gardner 1980). Second, the lithology and granulometry of the sediments varied among the studies.

Natural Sources of Turbidity/Sedimentation

In nature, high turbidity and rapid sedimentation are caused primarily by nearby river drainage, planktonic production, and resuspension of bottom sediments. In the following case histories, sedimentation effects were inferred by correlating coral parameters at various sites with corresponding values of turbidity or sedimentation.

Fanning Atoll Lagoon--

Roy and Smith (1971) studied the effects of turbidity on coral reef development in unpolluted Fanning Lagoon which is located in the central Pacific Ocean about 1,500 km south of Hawaii. The lagoon is characterized by areas of turbid water, with a calcium carbonate suspended load of 3.5 mg/l, and areas of clear water, where suspended solids averaged about 1.0 mg/l. Depositional rates for these areas were estimated at greater than 1.0 mm/yr and about 0.3 mm/yr, respectively. Because of the shallowness of the lagoon [mostly 4-15 m (13-49 ft)], illumination at the bottom was always greater than 5 percent incident light; i.e., well above the minimum light intensity required for coral growth.

Roy and Smith (1971) concluded that reefs in the clear water were ecologically different from those in the turbid water. Live corals covered about 60 percent of the bottom in clear water and about 30 percent of the bottom in turbid water (Roy and Smith 1971). Ramose corals accounted for 55 percent of the individuals at the turbid site and only 10 percent of those in the clear-water area. Accordingly, the reef structure in clear water was massive and steep-sided, while in turbid water it had a gentler slope and more sediment accumulation. Despite clear ecological differences, species composition in the two areas was similar with only four species being excluded from the turbid-water area (Maragos et al., 1970).

Negro-Bank Reefs, Western Puerto Rico--

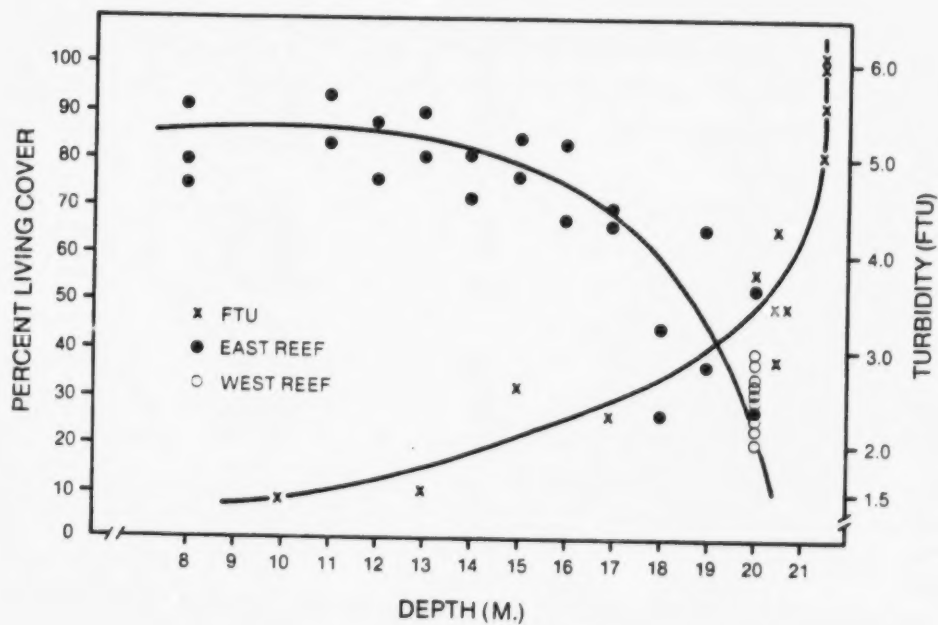
Loya (1976a) examined two reefs along the west coast of Puerto Rico to determine the role of turbidity and sedimentation in control of coral community structure. Coral diversity and living cover were high at the upper East Reef ($H' = 2.196$, cover = 79 percent), where average water turbidity and sedimentation were low (1.5 FTU and $3.0 \text{ mg cm}^{-2} \text{ day}^{-1}$, respectively). At the West Reef, species diversity and cover were relatively low ($H' = 1.830$, cover = 30 percent), while average turbidity and sedimentation were much higher (5.5 FTU and $15 \text{ mg cm}^{-2} \text{ day}^{-1}$, respectively). The major source of sedimentation stress appeared to be periodic resuspension and redeposition of fine sediments after heavy seas. Relationships between living coral cover, water turbidity, and water depth for the East and West Reefs are shown in Figure 6.

Species composition and the relative abundance of coral species were also influenced by sedimentation patterns. A coral community resistant to sedimentation was identified at the West Reef, with Montastrea cavernosa, Siderastrea radians, S. siderea, and Diploria strigosa being the most successful species. Although Montastrea cavernosa was dominant at both reefs, M. annularis (the main framework-builder throughout the Caribbean) was considered abundant only at East Reef.

Fouha and Ylig Bays, Guam--

Randall and Birkeland (1978) studied the effects of sedimentation on coral reefs of two bays in Guam. In both Fouha Bay and Ylig Bay, sedimentation was high near the river drainage at the head of the bay and low at the bay mouth (Table 7). Paralleling this decrease in sediment loading, the species richness and percent cover of coral communities generally increased along the shore-to-seaward gradient. Only data for the most favorable coral habitat (i.e., upper slope) are given in Table 6. Other environmental parameters, including temperature, pH, salinity, nitrates, and phosphates showed no systematic variation among stations.

Based on their data, Randall and Birkeland (1978) would expect a "depauperate coral community of less than 10 species covering less than 2



SOURCE: Loya, 1976a

Figure 6. Relationships between percent living coral cover, turbidity, and water depth, Negro-Bank Reefs, Puerto Rico.

TABLE 7. CORAL COMMUNITY STRUCTURE AND SEDIMENTATION RATES IN FOUHA AND YLIG BAYS, GUAM

Station	Sedimentation Rate mg cm ⁻² day ⁻¹	Coral Community - Upper Reef Slope		
		No. of Species	Percent Cover	Mean Colony Size (cm)
FN1	228	2	0.32	8.0
FN2	100	39	5.23	16.5
FN3	64.4	116	28.47	37.3
FN4	6.71	142	25.92	31.7
FS1	210	3	0	8.9
FS2	133	40	16.21	24.2
FS3	32.7	89	16.12	24.9
FS4	33.8	104	18.06	20.2
YN1	98.5	3	0	6.0
YN2	46.0	38	8.88	20.4
YN3	22.8	94	17.61	29.0
YN4	23.9	127	22.13	23.7
YS1	179	6	1.22	28.0
YS2	25.0	42	7.95	25.9
YS3	31.2	85	16.83	21.4
YS4	18.9	112	12.25	19.3

Note: Data are means of average sedimentation rates for eight consecutive 6-wk periods, as measured by tubular collectors approximately 2.4-cm diameter by 41-cm length. F = Fouha Bay, Y = Ylig Bay, N = north side of bay, S = south side of bay.

Source: Randall and Birkeland (1978).

percent of the solid substrate" where average sediment loads are about $160-220 \text{ mg cm}^{-2} \text{ day}^{-1}$. A "rich coral community of over 100 species covering over 12 percent of the solid substrate" is expected where average sedimentation rates are about $5-32 \text{ mg cm}^{-2} \text{ day}^{-1}$.

Discovery Bay, Jamaica--

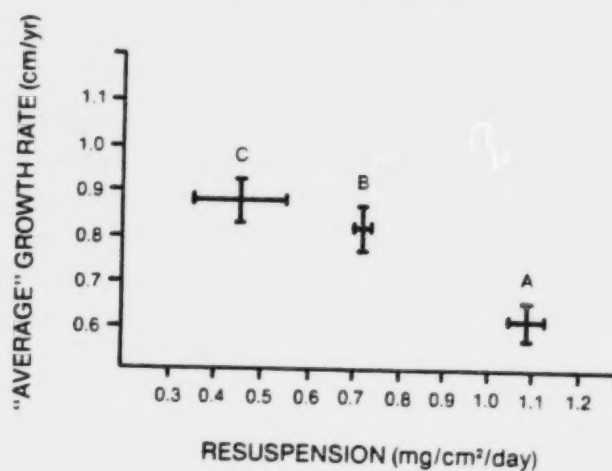
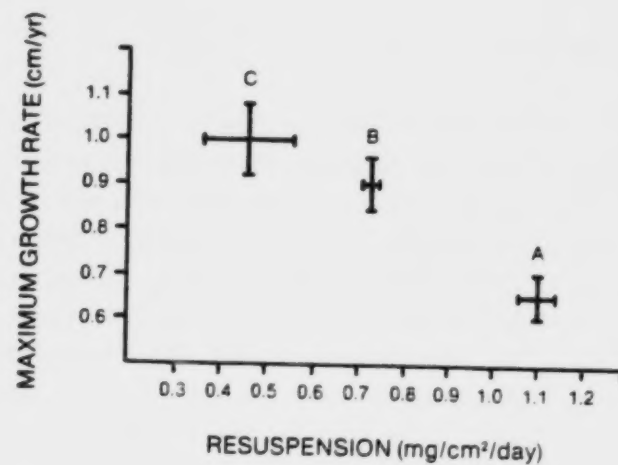
Dodge et al. (1974) and Aller and Dodge (1974) studied the growth of Montastrea annularis coral in relation to sediment resuspension in Discovery Bay. Both the average growth rate and the maximum growth rate of this species were inversely proportional to resuspension values (Figure 7). Resuspension rates in this case were measured by sediment traps placed 50 cm above the bottom, where the median particle size of resuspended material is less than 0.062 mm. "Resuspension values" also included fresh deposition, but Aller and Dodge (1974) showed that newly-deposited material was a minor component of the sediments collected in their traps.

Anthropogenic Sources of Turbidity/Sedimentation

Several of man's activities elevate turbidity and promote rapid sedimentation in coral reef environments. Anthropogenic sources include dredging operations, resuspension of sediments by boat traffic, and terrestrial runoff associated with urbanization or poor land management. Several case histories involving these activities are summarized below (also see Johannes 1975).

Castle Harbor, Bermuda--

Dredging was conducted during 1941-1943 in Castle Harbor, a semi-enclosed bay along the Bermuda coastline (Dodge and Vaisnys 1977). Increased sedimentation associated with the dredging operation was apparently responsible for decreased growth rates in corals. Impacts of substrate disruption persisted for several years after cessation of dredging, probably due to periodic resuspension of sediments. Eventually, all corals older than 20 yr (approximately 10 cm in height) died throughout the harbor. Population age structures in 1974 (35 yr after dredging) indicated that corals were still in a phase of high recruitment and recolonization.



Ranges are given for resuspension values.
 Error bars for coral growth rate equal
 $\pm$ two standard deviations.
 A, B, C are station names

SOURCE: Aller and Dodge, 1974

Figure 7. Growth rate of coral (*Montastrea annularis*) in relation to sediment resuspension rate, Discovery Bay, Jamaica.

As a result of the initial impact and extended recolonization period, dominance in the coral community shifted from Diploria strigosa to D. labyrinthiformis. Hubbard and Pocock (1972) demonstrated that D. labyrinthiformis is more capable of sediment rejection than D. strigosa, at least for particles larger than fine sand.

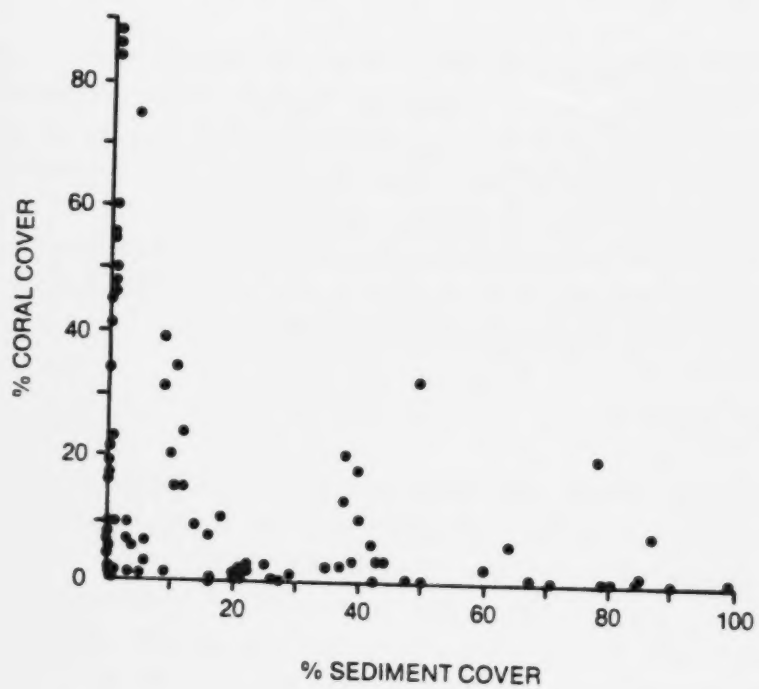
Piscadera Bay, Curacao--

Bak (1978) investigated the effects of dredging on a fringing reef community, 700 m (2,297 ft) from the dredge site. Increased turbidity during dredging resulted in a reduction of light intensity at 12-13 m (39-43 ft), from initial values of 27-30 percent to less than 1 percent of incident illumination. Colonies of Porites astreoides, an inefficient sediment rejector, lost their zooxanthellae and died. Calcification rates of other corals decreased by about 33 percent. Growth rate impacts appeared to persist for at least 30-60 days following dredging.

Kaneohe Bay, Hawaii--

Urbanization of the watershed surrounding Kaneohe Bay has resulted in extensive sediment influx, accounting for an average of about 1 m of sediment accumulation in the lagoon (Roy 1970; Laws 1981; Hollett and Moberly 1982). Fine, clay-sized sediments originating from terrestrial runoff have killed many corals, particularly in the southern basin and on shallow fringing reefs along the shore (Maragos 1972). The percentage of the substrate covered by living coral was inversely related to the percent cover by sediment (Figure 8). However, the relationship is not linear, and a threshold for development of abundant coral seems to exist at or below about 20 percent sediment cover. Above this value, most sites have less than 10 percent coral cover. Hollett and Moberly (1982) concluded that rapid shoaling of the southern bay was caused primarily by unrecorded dumping of dredge spoils and accidental spills, not by increased urbanization and altered land use.

By multivariate analysis of 25 environmental variables, Maragos (1972) found that light, salinity, and sewage variables (e.g., phosphate, dissolved oxygen variation, sediment cover) were the most important factors explaining



Note: Stations near sewer outfalls not included.

N = 91

SOURCE: Maragos, 1972

Figure 8. Coral cover versus sediment cover, Kaneohe Bay, Hawaii.

differences in growth rates of transplanted corals. Sedimentation rates accounted for less than 5 percent of the total variation in the growth rate data. However, sedimentation rates measured by Maragos (1972) were not representative of average conditions. Sediment traps were deployed for 45 days during a winter period of unusually heavy rain and terrigenous sediment influx. Coral growth rates were measured over a period of about 20 months.

Other Studies--

Additional case histories indicate a wide variety of coral reef responses to anthropogenic sediment influx, from essentially no impact (Griffin 1974; Sheppard 1980b; Dollar and Grigg 1981; Rogers 1982) to widespread degradation of reef communities (Brock et al., 1965, 1966; Marsh and Gordon 1974). For example, Griffin (1974) found no effects of dredging on a nearby coral patch reef in the Florida Keys. The dredge project resuspended 2,000 kg/day of sediments in the water column, an amount equivalent to 5 percent of the total baseline load for the entire study area. Sedimentation rates at the patch reef were not greatly enhanced by dredging, probably because the reef was located a considerable distance [0.7 km (0.5 mi)] upcurrent from the dredge site. Moreover, the coral patch reef was initially dominated by Siderastrea siderea, a species considered tolerant of high sedimentation and low DO (Griffin 1974). In contrast, turbidity and siltation from dredging operations adversely impacted over 7,000 acres of reef and lagoon near Johnston Island, with coral mortality in turbid waters ranging up to 40 percent (Brock et al., 1965, 1966).

SYNTHESIS

Sewage discharges may potentially impact coral reef communities through several processes:

- Nutrient enrichment
- Sedimentation
- Toxicity.

Adverse effects of sewage discharge on coral reefs have been observed primarily in poorly-flushed habitats; e.g., enclosed bays or lagoons. Significant adverse impacts of open ocean discharges have not been conclusively demonstrated for coral reef communities.

Only two comprehensive studies of sewage disposal into reef environments have been conducted; one in Kaneohe Bay (Hawaii) and one off the coast of southern Florida. In the former study, interpretation of sewage impacts is complicated by other anthropogenic perturbations; e.g., dredging and urban runoff (Banner 1974). Moreover, surveys conducted after sewage diversion from Kaneohe Bay covered an insufficient time for evaluation of the response of coral communities to decreased sewage inputs (Smith et al., 1981). In the other study off the coast of Florida, detailed results concerning sewage impacts are not yet available.

The following sections provide a synthesis of information on the impacts of sewage discharges on coral reef communities. Potential impacts are discussed in relation to the underlying mechanisms responsible for ecological change: nutrient enrichment, sedimentation, and toxicity. Data from case histories reviewed in previous chapters are integrated below to produce an overall assessment of sewage discharge effects.

NUTRIENT ENRICHMENT

Nutrient enrichment of coral reef communities produces a variety of direct and indirect effects. At low levels of nutrient input, primary production of benthic algae is enhanced without affecting biomass, species composition, or trophic structure (Kinsey and Domm 1974). Moderate levels of enrichment cause increased primary production and biomass in both phytoplankton and benthic algal populations (Laws and Redalje 1979; Smith et al., 1981). With increasing nutrient inputs, shifts in species dominance often lead to blooms of nuisance algae, especially planktonic flagellates (Clutter 1972; Mahoney and McLaughlin 1977) and benthic green or blue-green algae (Banner 1974; Snedaker, 24 August 1982, personal communication). Blooms of green algae (Ulva, Enteromorpha) and filamentous blue-greens have been observed near sewage outfalls on coral reefs in the Red Sea and in south Florida, respectively (Walker and Ormond 1982; Snedaker, 24 August 1982, personal communication). Some opportunistic algae are extremely sensitive to nutrient enrichment. For example, in Kaneohe Bay the green algae (Dictyosphaeria cavernosa) responded to sewage inputs at distances of over 10 km (6.2 mi) from the outfalls (e.g., Banner 1974). Enhanced plant growth and sewage particles around outfalls often attract fishes (Johannes 1975). However, algal blooms may persist even in the presence of enhanced grazer populations (Walker and Ormond 1982).

In addition to its direct effects on production and biomass of algae, moderate nutrient enrichment may lead to the following impacts:

- Enhanced bacterial populations may kill coral tissue (Mitchell and Chet 1975).
- Benthic algae may colonize coral skeletons, overgrow living corals, and form thick mats which kill all underlying organisms by blocking light and trapping sediment (Maragos and Chave 1973; Banner 1974; Walker and Ormond 1982).
- Elevated phytoplankton populations may reduce light penetration, which probably affects coral nutrition, growth, and survival through impacts on zooxanthellae (Smith et al., 1981).

- Increased water-column production may favor enhanced growth of benthic filter-feeders (e.g., sponges, bryozoans, tunicates), which outcompete corals for space (Maragos 1972; Birkeland 1977; Smith et al., 1981; Brock and Smith 1983).
- Opportunistic coral species such as Stylophora pistillata may dominate or replace other corals (Loya 1976c; Walker and Ormond 1982).
- Heterotrophic processes may overwhelm autotrophic production and calcification, leading to net erosion of reefs (Kinsey 1979).

The relative importance of the above mechanisms in controlling coral reef response to sewage inputs may vary among reef types and geographic regions. Unfortunately, the available data are insufficient to relate reef structure and species composition to response mechanisms.

Aside from the previously described impacts attributable to moderate levels of enrichment, extremely high nutrient inputs exert additional stress by promoting sedimentation and toxicity. High nutrient loading enhances planktonic primary production, which leads to increased sedimentation of organic material, e.g., in Kaneohe Bay (Smith et al., 1981). High phosphate levels also inhibit calcification by corals and coralline algae (Kinsey and Davies 1979).

The responses of coral communities to natural variations in nutrient loading provide interesting comparisons with enrichment effects due to sewage discharge. Birkeland (1977) described the effects of natural nutrient inputs by upwelling on coral reef communities off the Pacific coast of Panama. With a rich supply of nutrients, fouling organisms such as filamentous algae, bryozoans, and tunicates rapidly colonize open substrate and overgrow most coral recruits. As nutrient (and light) levels decrease, the rate of biomass accumulation on benthic substrates declines, and hermatypic corals have a better chance of reaching a size large enough to avoid being overgrown. Thus, r-selected fouling species dominate the later stages of reef succession in upwelling regions of the eastern Pacific. By

contrast, k-selected coral species dominate "climax" communities of the nutrient-poor Caribbean.

Moderate sewage inputs may mimic nutrient enrichment by natural upwelling. The parallels between sewage-modified communities and those of upwelling areas described by Birkeland (1977) are striking; e.g., rapid growth of benthic organisms, high biomass of filamentous algae, low diversity of corals, and domination by benthic filter-feeders.

The effects of sewage will vary with reef trophic status, which is often related to biogeographic differences. In nutrient-poor regions (e.g., central Pacific and Caribbean reefs), anthropogenic nutrient inputs may cause profound shifts in community structure; i.e., from domination by corals towards increased importance of r-selected filter-feeders. In upwelling areas (e.g., eastern-Pacific Panamanian reefs), moderate sewage inputs are less likely to cause dramatic changes since reef biota are already adapted to nutrient perturbations. Nonetheless, phosphate toxicity (Kinsey and Davies 1979) may play a role in determining reef calcification rates in upwelling areas.

SEDIMENTATION

Suspended solids in receiving waters for sewage discharges originate from three sources: particles contained in effluents, particulate organic matter produced by nutrient enrichment, and natural seston. The relative importance of these sources depends on wastewater treatment level.

Little information is available on the direct effects of sewage solids on hermatypic corals. In most sewage discharge studies (e.g., Russo et al., 1979; Smith et al., 1981; Walker and Ormond 1982), solids deposition appeared to be less important to coral reef status than nutrient enrichment. However, the case histories examined generally involved low mass emissions of sewage solids (e.g., secondary treatment at Kaneohe Wastewater Treatment Plant) or efficient dispersal of effluents (Mokapu Point outfall).

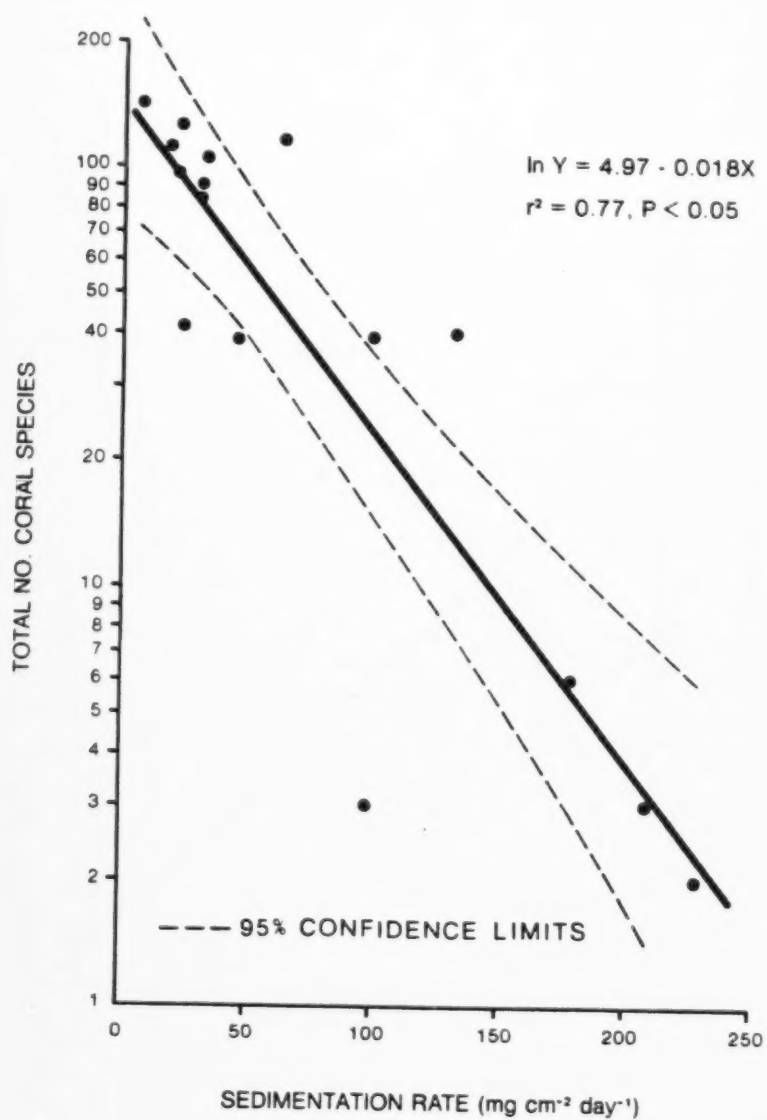
High turbidity and rapid sedimentation originate from sources other than sewage discharge (e.g., dredging, sediment resuspension, terrestrial runoff). Laboratory and field studies have indicated the following adverse impacts of suspended or deposited particles on corals:

- Death caused by burial (Edmondson 1928; Marshall and Orr 1931)
- Growth inhibition due to reduced light availability and additional energy expenditure for sediment rejection (Maragos 1972; Dodge et al., 1974; Bak 1978; Dallmeyer et al., 1982)
- Reduced coral abundance and species richness (Roy and Smith 1971; Loya 1976a; Randall and Birkeland 1978)
- Failure of coral recruitment (Maragos 1972; Dodge and Vaisnys 1977).

Few studies have examined long-term impacts of anthropogenic sedimentation on coral communities (e.g., Dodge and Vaisnys 1977).

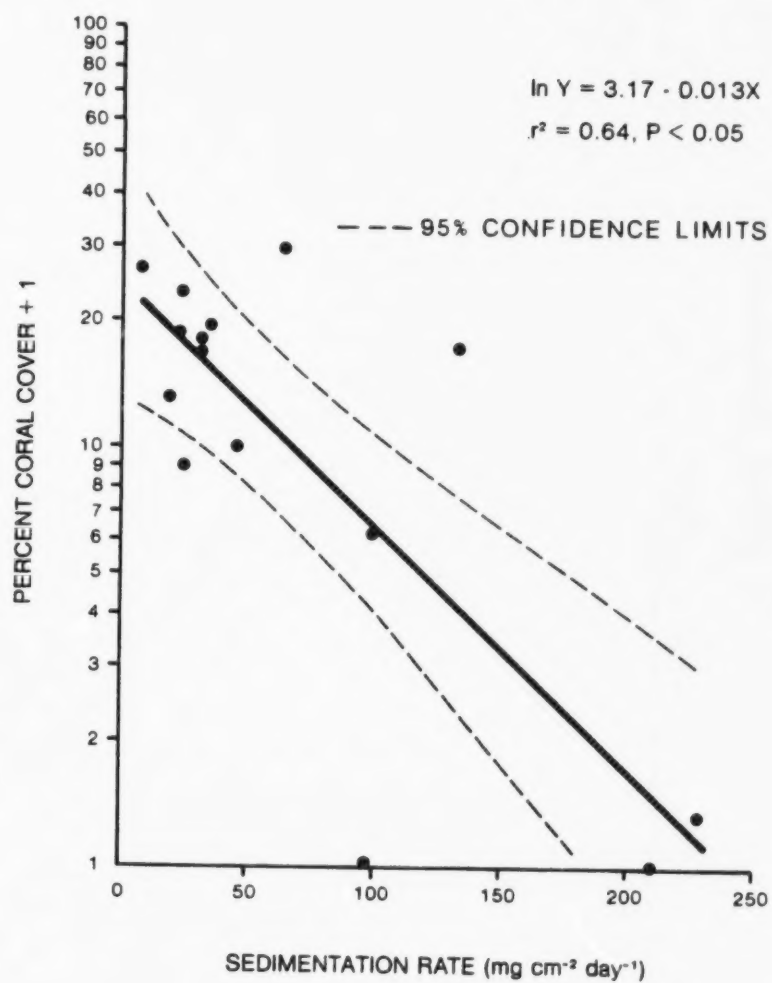
Despite the potential for adverse effects, most corals are capable of clearing a certain amount of sediment from the surface of the colony. An efficient sediment-rejector like Montastrea cavernosa is capable of cleaning itself at deposition rates up to about $14 \text{ mg cm}^{-2} \text{ day}^{-1}$ (Lasker 1980). Average sedimentation rates measured over extended periods (e.g., weeks, months) in natural coral reef habitats of the Caribbean zone range from 0.3 to $37 \text{ mg cm}^{-2} \text{ day}^{-1}$ (Griffin 1974; Ott 1975; Loya 1976a; Lasker 1980; Rogers 1982). In the Indo-Pacific region, corresponding sedimentation values range from 0.1 to $228 \text{ mg cm}^{-2} \text{ day}^{-1}$ (Marshall and Orr 1931; Smith and Jokiel 1975; Schuhmacher 1977; Randall and Birkeland 1978). At sediment deposition rates greater than $10\text{--}15 \text{ mg cm}^{-2} \text{ day}^{-1}$, coral communities exhibit apparent modifications in response to sedimentation (or turbidity) stress; e.g., appearance of sediment-resistant species, reduced abundance and diversity, predominance of branching growth forms.

Although quantitative data are limited, relationships between coral response parameters and sedimentation rate have been inferred by many researchers (e.g., Dodge et al., 1974; Loya 1976a; Randall and Birkeland 1978). Response curves based on data from Randall and Birkeland (1978) are presented here to illustrate the functional dependence of coral community structure on the rate of sediment deposition (Figures 9, 10, 11, and 12).



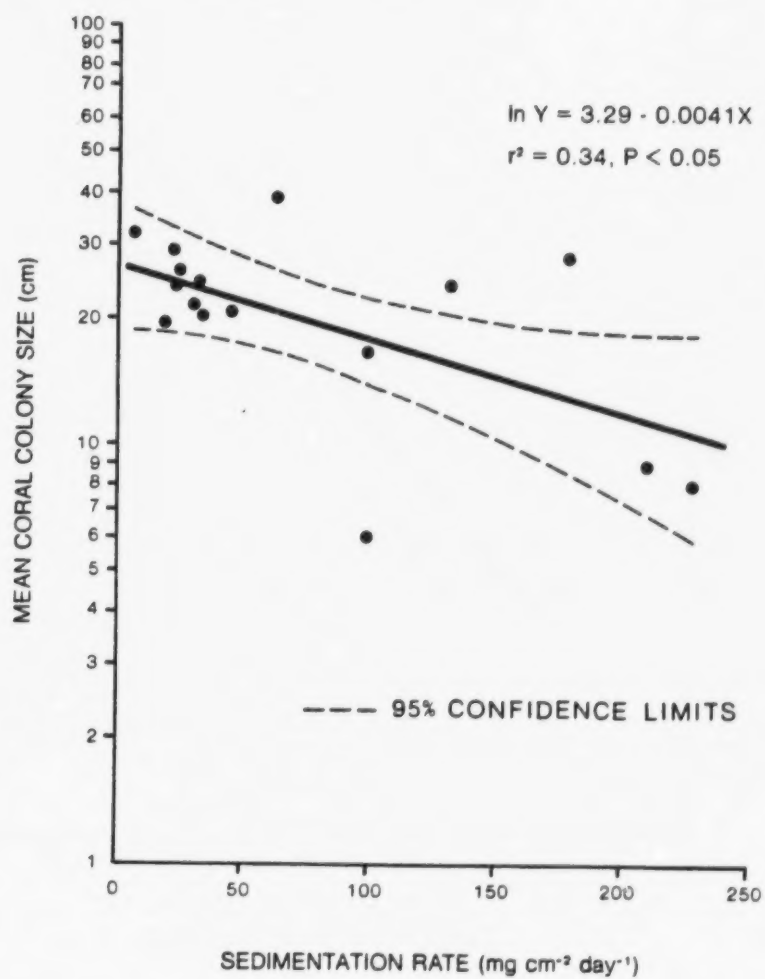
SOURCE Data from Randall and Birkeland, 1978

Figure 9. Coral species richness as a function of sedimentation rate, Guam.



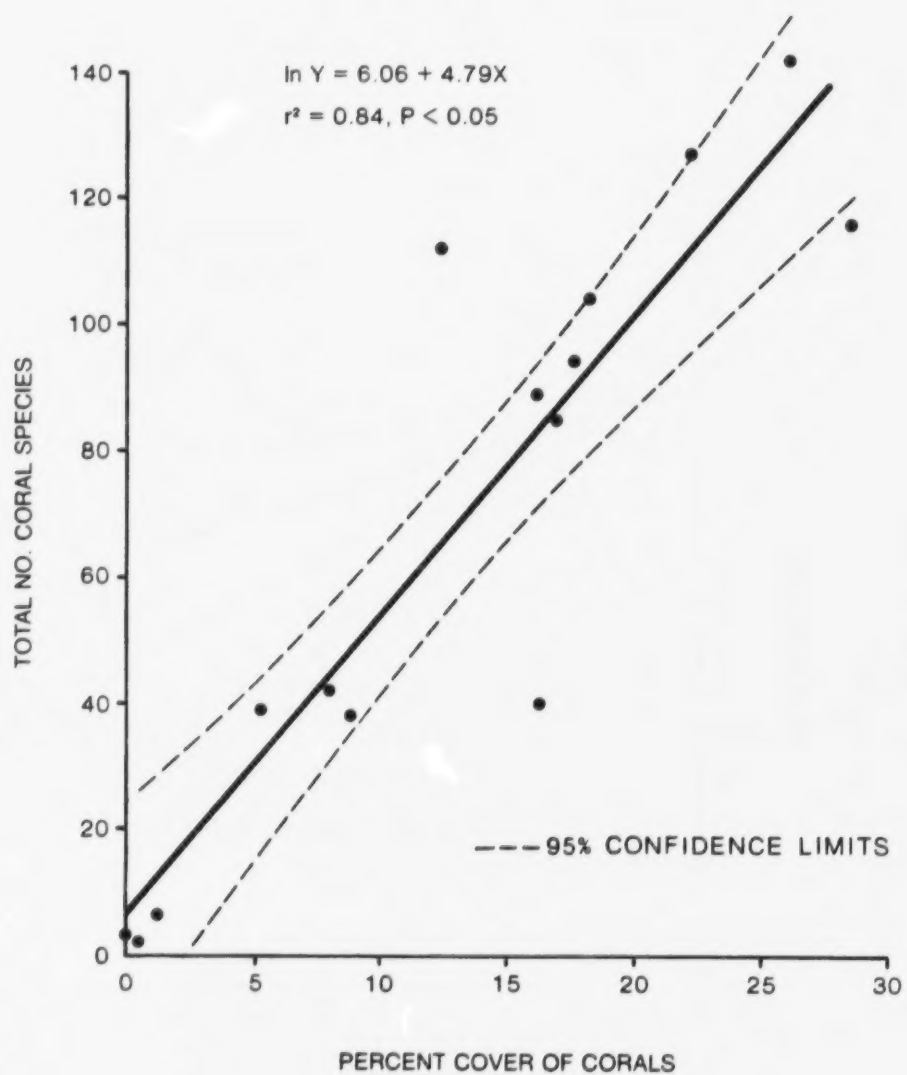
SOURCE: Data from Randall and Birkeland, 1978

Figure 10. Coral percent cover as a function of sedimentation rate, Guam.



SOURCE: Data from Randall and Birkeland, 1978

Figure 11. Coral colony size as a function of sedimentation rate, Guam.



SOURCE: Data from Randall and Birkeland, 1978

Figure 12. Coral species richness as a function of coral cover, Guam.

Only the data from the upper slope communities were used to develop the curves in Figures 9 to 12. Because the upper slope communities form the best-developed reefs and are normally subject to the least sedimentation relative to other coral habitats, they should reveal the clearest responses to gradation of sediment influx. Lines were fitted to the data in Table 6 by using least-squares linear regression. Species richness, percent cover, and mean colony size of corals are each inversely related to sedimentation rate. Coral species richness is positively correlated with percent cover ($r^2 = 0.75$; $p < 0.05$), indicating that no single species is a clear competitive dominant (Porter 1974). In general, the response of coral communities in other reef zones to sediment inputs is similar to the responses illustrated here.

The variability of coral community parameters (species richness, percent cover, and mean colony size) changes along the gradient of sedimentation rates shown in Figures 9, 10, and 11. At sedimentation values less than $40\text{--}50 \text{ mg cm}^{-2} \text{ day}^{-1}$, the variance around the regression lines is relatively low. Since many corals of the upper reef slope flourish only where sedimentation is minimal, the low variability of species number and percent cover at sedimentation rates less than $40\text{--}50 \text{ mg cm}^{-2} \text{ day}^{-1}$ may reflect the control of community structure through biological mechanisms such as competition, predation, and mutualism. At the highest sedimentation rates ($150\text{--}250 \text{ mg cm}^{-1} \text{ day}^{-1}$), the low variance in community structure is a reflection of environmental stress. Only a few resistant species are able to survive the extreme sediment influx, and community structure is directly controlled by physical variables rather than biological interactions. In the midrange of sedimentation values, both physical and biological mechanisms may be important in determining community structure, with moderate stress producing a number of alternative community structures. Thus, the variance in community structure is greatest at the midrange of sediment stress ($50\text{--}150 \text{ mg cm}^{-2} \text{ day}^{-1}$). Although the data obtained from Randall and Birkeland (1978) may be too variable for precise prediction of coral reef impacts following changes in the sedimentation regime, they nonetheless serve to illustrate the type of data required for development of quantitative response models.

Based on information from Randall and Birkeland (1978) and other case histories reviewed previously, an impact scale was developed for various levels of sediment deposition (Table 8). Individual coral communities were

TABLE 8. ESTIMATED DEGREE OF IMPACT ON CORAL COMMUNITY CAUSED
BY VARIOUS LEVELS OF SEDIMENTATION - SUMMARY

Sedimentation Rate mg cm ⁻² day ⁻¹	Estimated Impact
1-10	SLIGHT TO MODERATE: ● Decreased abundance/cover ● Altered growth forms ● Decreased growth rates ● Possible reductions in recruitment ● Possible reductions in numbers of species
10-50	MODERATE TO SEVERE: ● Greatly decreased abundance/cover ● Greatly decreased growth rates ● Predominance of altered growth forms ● Reduced recruitment ● Decreased numbers of species ● Possible invasions of opportunistic species (e.g., algae)
> 50	SEVERE TO CATASTROPHIC: ● Severe degradation of communities ● Most species excluded ● Many to most colonies die ● Recruitment severely reduced ● Regeneration slowed or stopped ● Invasion of open substrates by opportunists ● Coral cover severely reduced

rated according to their response to sedimentation (i.e., slight to moderate, moderate to severe, etc.) and the corresponding range of sedimentation rates was recorded under the appropriate response category (Table 9). The levels of sedimentation listed in Table 8 represent a summary of the case history data compiled in Tables 3 and 9. Because of the limited data available, the degree of impact for each level of sediment deposition should be considered tentative.

A preliminary assessment of sedimentation impacts associated with sewage discharges has been conducted as part of the technical review process for Section 301(h) (Clean Water Act) applications received during 1979. Maximum predicted deposition rates for sewage outfalls located in Hawaii and Puerto Rico (regions with coral reefs) are summarized in Table 10. It should be emphasized that the deposition rates in Table 10 are "worst-case" estimates for sewage solids accumulation; i.e., they do not incorporate resuspension and transport of sediment particles by waves and currents. However, they also do not include the effects of toxic substances in sewage and ambient sediment influx. None of the proposed outfalls will discharge directly to a coral reef habitat. Despite these limitations, it is interesting to compare the sediment deposition rates predicted for these sewage discharges (Table 10) with the ranges of sedimentation rates derived for various levels of impact (Table 8). All of the sediment deposition rates calculated for the proposed sewage discharges are extremely low compared to natural sedimentation rates found on most coral reefs.

TOXICITY

Toxic effects on corals and other reef organisms may result from one or more of the chemicals commonly found in sewage effluent: metals, chlorine, phosphate, pesticides, and petroleum hydrocarbons. Concentrations of these chemicals vary greatly among discharges and through time for a specific discharge. Because a multitude of toxic substances may occur in sewage effluent, additive and synergistic effects of pollutants are important. Aside from the effects of petroleum hydrocarbons, toxic effects on corals and other reef organisms have received little attention.

The effects of free residual chlorine on reef organisms were reviewed by Johannes (1975). He cites several studies which indicate that reef

TABLE 9. ESTIMATED DEGREE OF IMPACT ON CORAL COMMUNITY
CAUSED BY VARIOUS LEVELS OF SEDIMENTATION

Estimated Impact ^a	Sedimentation Rate mg cm ⁻² day ⁻¹	Reference
Slight to moderate	6-20	Randall and Birkeland (1978)
	3	Loya (1976a)
	7-8	Lasker (1980)
	~1	Dodge et al. (1974)
	1-15	Ott (1975)
Moderate to severe	20-50	Randall and Birkeland (1978)
	15	Loya (1976a)
	14	Lasker (1980)
	37 (6-125)	Griffin (1974)
	10	Morelock et al. (1979)
Severe to catastrophic	>50	Randall and Birkeland (1978)

^a Categories correspond to impact levels identified in Table 8.

TABLE 10. WORST CASE ESTIMATES OF SOLIDS DEPOSITION RATES FOR
SEWAGE OUTFALLS NEAR CORAL REEFS^a

Outfall Location	Average Sedimentation Rate mg cm ⁻² day ⁻¹
Honouliuli, HI	0.036 ^b
Sand Island, HI	0.029 ^c
Kailua-Kaneohe, HI	0.056 (Area I)
Waianae, HI	0.003 (Sector I)
Ponce, PR	0.011
Guayanilla, PR	0.006
Humacao, PR	N 0.004 S 0.005 E 0.012 W 0.028
Aguadilla, PR	NE 0.013 SE 0.001 NW 0.002 SW 0.005

^a Predicted maximum deposition rate for total sewage solids (inorganic and organic) during 90-day critical period of high mass emissions.

^b Honouliuli estimate, represents a 36-day period of onshore currents.

^c Sand Island estimate represents a 21-day period of onshore currents.

fishes are sensitive to chlorine. Davis (1971 as cited by Johannes 1975) tested the effects of 0.49 mg/l chlorine on the planulae of three species of Hawaiian corals and found that exposure for up to 7 hours was not lethal. Adult corals were not tested.

Phosphate pollution has been implicated as a factor contributing to the decline of reef ecosystems in Eilat, Red Sea (Fishelson 1973; Loya 1975, 1976b; Fishelson 1977). However, chronic oil pollution may account for most, if not all, of the pollution damage observed on the Eilat reef flat (Rinkevich and Loya 1977; Loya and Rivkevich 1979). Effects of elevated phosphorus levels should not be dismissed, however. Kinsey and Davies (1979) experimentally enriched a patch reef at One Tree Island, Great Barrier Reef, with phosphate (2 μ M) and nitrogen (20 μ M urea plus ammonia) for 3 hours each day at low tide over a period of 8 months. They found that reef calcification was reduced by at least 50 percent, and attributed this suppression to phosphate enrichment.

Reimold (1975) sampled coastal biota from Puerto Rico and the Virgin Islands to determine tissue levels of chlorinated hydrocarbons (dieldrin, DDT, DDE, TDE, PCBs) and mercury. Body burdens of these pollutants were detected at low levels in reef fishes and invertebrates at one or more of the 15 survey sites. Olafson (1978) surveyed organochlorine pesticide body burdens in two species of fish, two species of corals, and one species of bivalve mollusc from the Great Barrier Reef. Lindane was the only organochloride consistently detected, and tissue levels were very low. Both authors hypothesized that land-use practices were a major factor affecting the distributions of toxic substances in the biota.

Neither Reimold (1975) nor Olafson (1978) assessed the effects of toxic residues in reef organisms. To date, only the study of McCloskey and Chesher (1971) has examined the effects of chlorinated hydrocarbons on coral colonies. McCloskey and Chesher (1971) subjected the corals Montastrea annularis, Acropora cervicornis, and Madracis mirabilis to mixtures of p, p'-DDT, dieldrin, and Aroclor 1254 (a PCB) in equal proportions. Colonies were dosed with 10, 100, and 1,000 ppb of each of the three compounds. No changes in feeding behavior, polyp extension, sediment clearing, settling of coral associates, or crystal formation were observed. However, the authors recorded an increase in respiration (R) and a decrease

in photosynthesis for all three species, such that the P/R ratio fell below 1.0. Photosynthesis remained depressed for up to 4 days, at which time the experiment was terminated.

Although petroleum hydrocarbons are not usually present at high concentrations in sewage effluents, chronic exposure of reef organisms to low concentrations of these toxicants is a potential problem. Loya and Rinkevich (1980) thoroughly reviewed the effects of oil pollution on reef communities and documented a multitude of detrimental impacts, many of which exhibited delayed response. Researchers have documented decreased viability of coral colonies (Eisler 1975; Loya 1975, 1976b; Rinkevich and Loya 1977, 1979; Peters et al., 1981), decreased growth rates (Birkeland et al., 1976), damage to cells and tissues (Birkeland et al., 1976; Peters et al., 1981), altered behavior (Eisler 1975; Reimer 1975a, 1975b; Cohen et al., 1977; Loya and Rinkevich 1979), and excessive production of mucous with a rich bacterial flora (Mitchell and Chet 1975; Ducklow and Mitchell 1979). Elevated bacterial populations were implicated as contributors to coral demise (Mitchell and Chet 1975; Ducklow and Mitchell 1979). Loya and Rinkevich (1980) also documented effects of oil pollution on the reproduction and recruitment of corals. Observed effects included reduced colonization by corals in areas subjected to chronic oil pollution (Loya 1975, 1976b; Rinkevich and Loya 1977), deleterious effects on the reproduction systems of corals (Loya 1975; Rinkevich and Loya 1977, 1979; Peters et al., 1981), and premature release of coral planulae upon contact with petroleum compounds (Cohen et al., 1977; Loya and Rinkevich 1977, 1979). Loya and Rinkevich (1979) speculated that premature release of planulae results in reduced survival and settlement, especially if the receiving environment is contaminated with hydrocarbons.

In addition to toxic substances in the sewage effluent, toxic substances associated with enriched bottom sediments may also have an important influence on coral reef communities near sewage outfalls. Hydrogen sulfide in bottom sediments increases with proximity to sewage discharge sites (Sorokin 1973c). The release of toxic sulfides from bottom sediments in southern Kaneohe Bay may have been responsible for coral mortality and the low diversity of reef communities (Maragos 1972, Maragos and Chave 1973). In some instances, resuspension of sediments during storms might also result in a transfer of toxic materials (e.g., PCBs, metals,

pesticides) into the water column, increasing the likelihood of biological contamination.

Past studies have addressed only the short-term effects of toxic substances on coral reef organisms. Yet results of several studies of petroleum contamination suggest that sublethal and lethal effects may, in some cases, be delayed or prolonged. Although uptake of petroleum hydrocarbons by corals occurs relatively quickly, depuration does not. Peters et al. (1981) and Knap et al. (1982) reported that depuration of corals did not occur within 2 weeks after cessation of exposure. Death of coral tissue following short-term exposure to Bunker C oil may not occur until 2 weeks after exposure (Birkeland et al., 1976). The persistence of delayed toxic effects and the severity of those delayed effects have yet to be established for coral reef systems.

CONCLUSION

The response of coral reef communities to sewage effluent components, including nutrients, suspended solids, and toxic substances may vary from reef to reef depending on the specific reef biota and biogeographic zone. Spatial and temporal variations in the response of coral reefs to sewage inputs and the mechanisms underlying these responses are not fully understood. Available information indicates, however, that most coral species can tolerate limited amounts of nutrient enrichment and sedimentation rates. Within the range of environmental variation observed in nature, coral reef communities exhibit dramatic shifts in structure and function along gradients in nutrient influx (e.g., from oligotrophic seas to eutrophic upwelling areas) and sedimentation rates (e.g., from inner bays and lagoons to offshore reef slopes).

Stimulation of marine productivity by slight nutrient enrichment may be viewed as a potential benefit, but possible subtle changes in reef ecology due to moderate sewage inputs may have chronic or long-term impacts on these distinctive habitats of limited distribution. Alterations of coral-reef community composition, diversity, and abundance caused by moderate sewage inputs are similar to natural variations of reef communities along environmental gradients. However, high nutrient inputs from sewage discharges are potentially harmful to coral reef communities. Although most coral species can tolerate limited eutrophication and sedimentation, the cumulative impacts of a large discharge in poorly-flushed waters may lead to disruption of coral community structure and eventual erosion of the reef through mechanical, chemical, and biological processes. Climax species of corals, i.e., slow growing, slow reproducing, K-selected forms, may be more sensitive to sediments and toxic chemicals contained in sewage effluents than pioneering (r-selected) species. Moreover, high nutrient inputs may enhance the growth of pioneering species causing a shift in species composition eventually, leading to the replacement of corals by other benthic species (e.g., bryozoans, tunicates, and filamentous algae).

Available data suggest that observed impacts of nutrient enrichment in coral reef habitats are greater than impacts of sewage-solids deposition. Maximum predicted sedimentation rates near open-coast outfalls discharging primary-treated effluent are one to two orders of magnitude lower than natural sedimentation rates on biologically-rich coral reefs (e.g., $0.001-0.056 \text{ mg cm}^{-2} \text{ day}^{-1}$ for sewage solids vs. $1-10 \text{ mg cm}^{-2} \text{ day}^{-1}$ for natural sedimentation).

Despite the large amount of information available on the basic ecology of coral reefs, current data on anthropogenic impacts are limited. Although specific information is available for certain localities, at present, data gaps exist in the following key areas of concern:

- Effects of toxic chemicals in sewage effluents
- Importance of toxicity, nutrient enrichment, sedimentation, and oxygen depression as a function of sewage loading and flushing potential
- Synergistic and additive effects of various components in sewage effluents
- Recovery times from sewage impacts.

APPENDIX: LITERATURE SEARCH AND INFORMATION SOURCES

An extensive file of information on coral reef ecology and impacts of sewage was already on hand in the Tetra Tech library. This file was updated in three ways:

- Computerized search of biological information
- Manual library search
- Personal contacts with recognized experts on coral reef ecology.

COMPUTERIZED SEARCH

A computerized search for literature on pollution of coral reefs was conducted by CERL (U.S. EPA, Corvallis, OR) during March, 1982. The following data bases were accessed: BIOSIS, TOXLINE, NTIS, and WRA. From computer printouts of references, Henry Lee and Mark Schaefer of CERL compiled a "Bibliography of Publications related to the Effects of Sewage Discharges on Coral Reefs (July, 1982)". This bibliography, composed of 114 references, was made available to Tetra Tech.

MANUAL LIBRARY SEARCH

A manual search of recent journal and abstract issues was conducted from August 12, 1982, through December 1, 1982, using the library facilities of the University of Washington. In general, journal issues for the years 1981 and 1982 were examined. Earlier issues were examined only for high priority items, some of which are not published annually (e.g., Proc. Int. Coral Reef Symp.). The search included the following journals.

Adv. Ecol. Res.
Amer. Natur.

Aquat. Bot.
 Austr. J. Mar. Freshwat. Res.
 Biol. Bull.
 Biol. Conserv.
 Biol. Geol. of Coral Reefs
 Biol. Oceanogr.
 Bull. Mar. Sci.
 Coral Reefs
 Ecology
 Environ. Conserv.
 Environ. Pollut.
 Est. Coastl. Mar. Sci.
 Est. Coastl. Shelf Sci.
 J. Conseil Explor. Mer
 J. Exp. Mar. Biol. Ecol.
 J. Mar. Res.
 Limnol. Oceanogr.
 Mar. Biol.
 Mar. Ecol. Prog. Ser.
 Mar. Environ. Res.
 Mar. Pollut. Bull.
 N. Zealand J. Mar. Freshwat. Res.
 Oceanogr. Acta
 Oecologia
 Oikos
 Pac. Sci.
 Proc. Int. Coral Reef Symp.
 Science
 Water Air Soil Pollut.

In addition, the following reference sources were consulted:

Aq. Sci. Fish. Abstr. 1982 12(1-3)
 Biol. Abstr. 1982 74(1-8)
 Current Contents 1982 June 7-October 18
 Oceanic Abstr. 1982 19(1-3)
 1981 18(1-6)

PERSONAL CONTACTS

Personal contacts with recognized experts on coral reef ecology were made largely by phone and/or letter. The following individuals were contacted:

<u>Name</u>	<u>Affiliation</u>
Dr. C. Birkeland	Univ. Guam Marine Lab., Guam
Dr. D. Cheney	Shapiro and Associates, Seattle, WA
Dr. S. Coles	Hawaiian Electric Co., Honolulu, HI
Dr. C. Cutress	Univ. Puerto Rico, Mayaguez, PR
Dr. T. Duke	U.S. EPA, Gulf Breeze, FL
Dr. E. Gomez	Univ. Philippines, Quezon City, Philippines
Dr. J. Gonzales	Univ. Puerto Rico, Mayaguez, PR
Dr. G. Griffin	Univ. Florida, Gainesville, FL
Dr. R. Highsmith	Friday Harbor Lab., San Juan Is., WA
Dr. W. Jaap	Florida Dept. Natural Resources, St. Petersburg, FL
Dr. R. Johannes	CSIRO, W. Australia
Dr. P. Jokiel	Hawaii Institute Marine Biology, Kaneohe, HI
Dr. D. Kinsey	Univ. Georgia, Athens, GA
Dr. A. Kohn	Univ. Washington, Seattle, WA
Dr. J. Kumagai	M & E Pacific, Inc., Honolulu, HI
Dr. H. Lasker	State Univ. New York, Buffalo, NY
Dr. J. Maragos	U.S. Army Corps of Engineers, Kaneohe, HI
Dr. V. McFarland	Waterways Experiment Station, Vicksburg, MS
Dr. E. Powell	Texas A & M Univ., College Station, TX
Dr. R. Randall	Univ. Guam Marine Lab., Guam
Dr. C. Rogers	W. Indies Lab., Farleigh, Dickinson Univ., USVI
Dr. E. Shinn	U.S. Geological Survey, Miami Beach, FL
Dr. T. Sleeter	Bermuda Biological Station Research, Bermuda
Dr. S. Smith	Hawaii Institute Marine Biology, Kaneohe, HI
Dr. S. Snedaker	Univ. Miami, Miami, FL
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Dr. F. Talbot	California Academy Science, San Francisco, CA
Dr. R. Tsuda	Univ. Guam Marine Lab., Guam

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